Design of Effective Riparian Management Strategies for Stream Resource Protection in Montana, Idaho, and Washington

Technical Report #7

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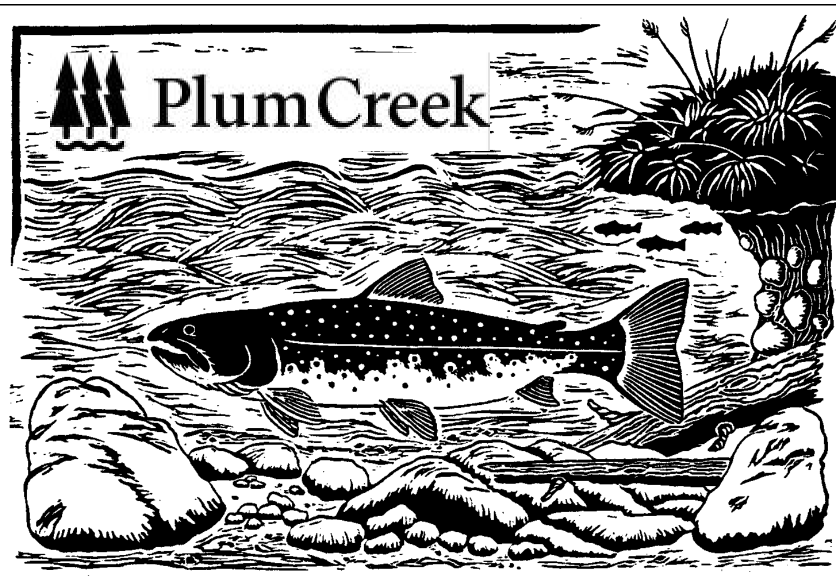
Plum Creek Timber Company



Native Fish Habitat Conservation Plan

DESIGN OF
EFFECTIVE RIPARIAN MANAGEMENT STRATEGIES FOR STREAM RESOURCE PROTECTION IN MONTANA, IDAHO, AND WASHINGTON

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NATIVE FISH HABITAT CONSERVATION PLAN
TECHNICAL REPORT No. 7

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TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

1.0 INTRODUCTION

An important element of contemporary forestry is to determine appropriate levels of management in riparian areas. From the perspective of the fisheries resource, riparian areas play an important role in the structure and function of the stream ecosystem (Meehan 1991, USDA et al. 1993). From the perspective of land owners, timber in riparian areas provides an important economic resource. These competing values create a management dilemma that is not easily solved (Hairston-Stand and Adams 1997). A successful and durable solution to this dilemma strikes a balance between resource protection and resource use in riparian areas. We believe a balanced solution lies in a better understanding of the degrees to which management actions affect fish or fish habitat, and in identification of the locations and conditions where differences occur. With this foundation established, management strategies can be developed to address and accommodate both the ecologic and economic value of riparian areas.

1.1 The Role of Riparian Areas in Supporting Native Salmonid Fish Populations

Riparian areas exert a strong influence on the structure and function of stream ecosystems, which in turn support an array of native salmonid fish species. Riparian influences can be subdivided into two main categories: physical and biological. The physical influences derive mainly from vegetative structure provided by trees, shrubs, and grasses. These structural influences principally affect channel morphology and habitat diversity (i.e., complexity), the stream's thermal regime, and the stream's sediment regime.

The biological influences of riparian areas are similarly important though less readily obvious. Riparian vegetation helps govern the flux of organic matter to and through the stream (Murphy and Meehan 1991). The flow of nutrients from surface sources (e.g., leaf litter, terrestrial insects) is the primary process, but riparian condition may also affect sub-surface sources (e.g., hyporheic mineral nutrients and aquatic insects (Ward and Stanford 1989). In-channel large woody debris (LWD) can contribute to nutrient cycling by retaining organic matter (Murphy and Meehan 1991), including carcasses of post-spawning anadromous salmonids (Cederholm et al. 1989, Bilby et al. 1996).

All portions of the drainage network contribute in some way to the creation or maintenance of a stream environment suitable for salmonids. However, not all stream reaches contribute equally. In most watersheds of the western United States, only a fraction of the entire stream system is inhabited by salmonids. The unoccupied perennial and intermittent streams are interwoven with watershed processes; these reaches ultimately transport sediment, water, and nutrients to fish-bearing reaches. Riparian areas along these stream reaches function in concert with other watershed processes (e.g., mass wasting) to influence the amount and timing of the materials that are conveyed. Even within stream reaches occupied by fish, there is tremendous variability in the type and quality of fish habitat, depending on the location of the reach within the drainage network (Montgomery and Buffington 1997, Vannote et al. 1980, Watson and Hillman 1997) and its geographic location (Platts 1979, Watson and Hillman 1997).

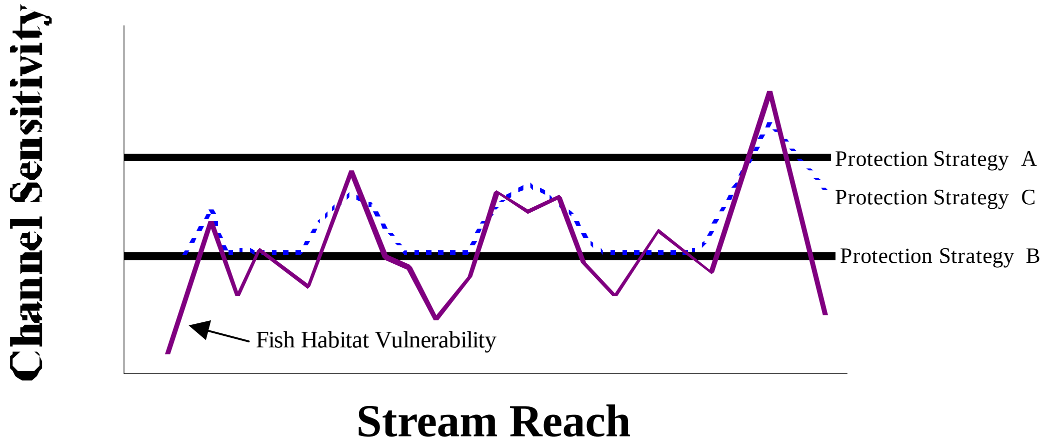
Separate from the innately different habitat-supporting roles played by distinct reaches of the channel network are the relative effects forest management activities can have on watershed processes in each reach. This relative influence of management activities on watershed processes may differ depending on the geographic location of a watershed, and where an activity within a watershed is conducted. For instance, geographic regions may vary in the magnitude or types of unstable slopes (e.g., eastern vs. western Washington). Unstable slopes are often localized within watersheds. Therefore, it is often difficult to determine the impact of timber management activities without first knowing the spatial and temporal scope and scale of the intended actions.

1.2 The Influence Of Timber Management On Riparian Areas and Stream Ecosystems

Riparian structure and ecological function can be negatively influenced by improper timber and livestock management (Meehan 1991, Elmore and Beschta 1987, Platts 1990). The degree of these influences has only recently been recognized. The role of riparian areas in supporting fish populations has long been acknowledged (Marsh 1864), but only recently have management activities adjusted to accommodate this role. Prior to the 1970s, clearcut timber harvest to the edge of the stream was common, as was log yarding or construction of roads in stream channels. Splash damming, intensive and unrestricted cattle grazing, and deliberate woody debris removal programs were routine. A review of these historical practices within the project area is presented by Sugden and Light (1998), and McGreer et al. (1998). Since the adoption and implementation of forest practices rules in Washington (1974), Idaho (1974) and Montana (1989), and their subsequent revisions, the most catastrophic and detrimental practices have largely ceased or are greatly attenuated. Riparian protection measures are now an important element of all federal, state, and local rules and regulations governing forest management activities.

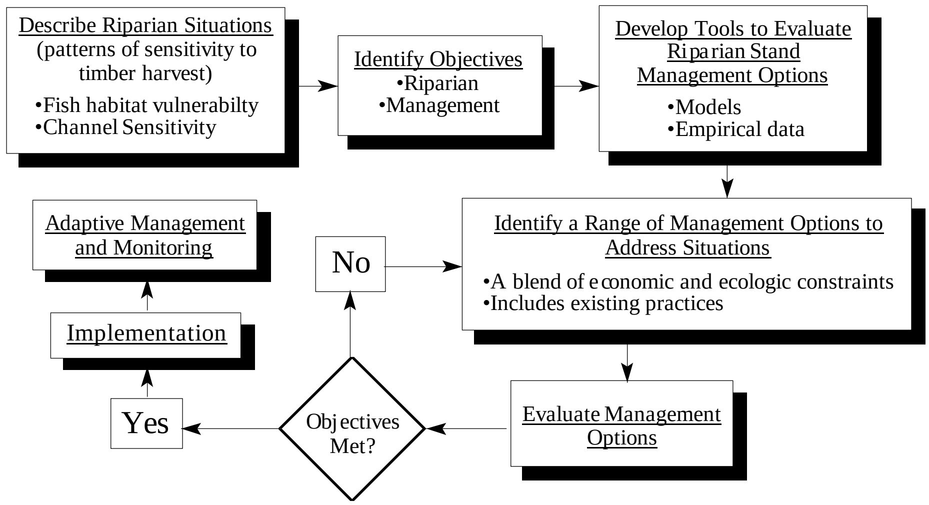
In the States of Washington, Idaho, and Montana, riparian-related protection measures contained in forest practice rules and regulations are driven by a general desire to protect fish habitat and water quality. In Washington, a process called Watershed Analysis is used to modify, adjust, and/or replace the general rules to fit watershed- or reach-specific conditions (Washington Forest Practices Board 1995; see Sugden and Light 1998 for further details). Federal programs such as the Aquatic Conservation Strategy in the President's Forest Plan (USFS 1994), PACFISH (USDA and USDI 1995), and INFISH (USFS 1996) were more specifically crafted to respond to dwindling fish stocks (Nehlson et al. 1990) and riparian-dependent wildlife (USDA et al. 1993). Like state regulations, the federal management strategies were developed for application over broad geographic areas (e.g., all federal lands in the Pacific Northwest). A federal assessment methodology modeled after Washington's Watershed Analysis process (called "Ecosystem Analysis at the Watershed Scale") allows for, but does not specifically guide, development of site-specific adjustments to default riparian prescriptions. We seek to improve upon the speed and degree of fine-tuning of riparian management enabled by Watershed Analysis.

Figure 1. Generalized depiction of the natural variability in fish resource sensitivity according to location in the drainage network, and the ability of three riparian management approaches to address this variability.



The degree to which existing state and federal riparian management systems provide protection for stream resources is illustrated in Figure 1. Federal riparian protection strategies are represented by strategy "A." They prescribe comparatively wide, no harvest zones throughout the drainage network (e.g., USDA 1994). This type of prescription is simple, requires little information to implement, assumes little risk, and is therefore least flexible and most restrictive from the perspective of commercial timber use. In contrast, state forest practice rules governing private lands assume somewhat greater risk in consideration of the commercial interests of private landowners (strategy "B"). The degree of increased risk depends on the type of practice and the sensitivity of watershed processes to that practice. In many cases, state rules are adequate to address a particular sensitivity, while in other cases they fall short. An ideal system would "shrink-wrap" management actions around channel sensitivity. However, limits to our understanding of watershed structure and function, the costs of gathering the necessary information, and the costs of applying complex practices prevent us from achieving this perfect outcome. Instead, we strive for the best possible fit, as depicted in strategy "C."

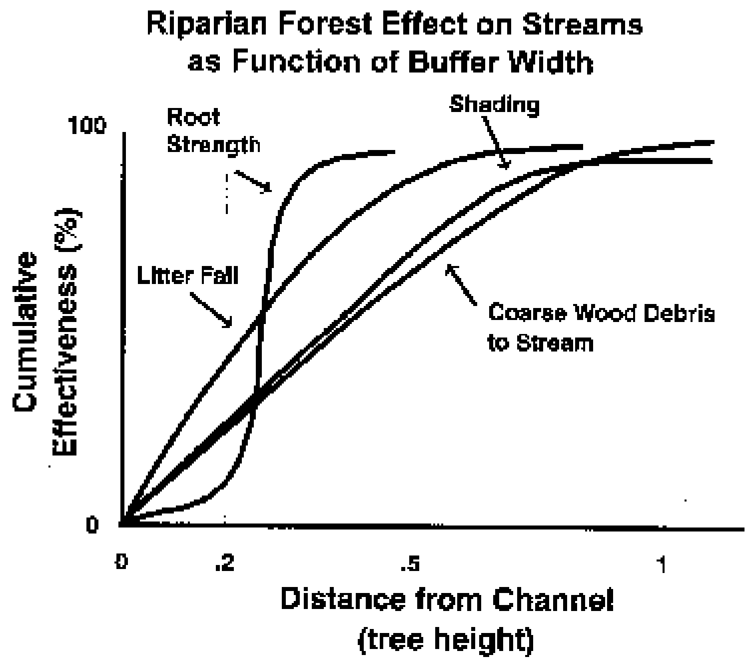
Figure 2. Process for developing riparian management actions that respond to patterns of fish habitat vulnerability and channel sensitivity within and among watersheds.



The practical question then, is how to achieve the balance between protection and sensitivity? What information do managers need to accurately recognize and respond to patterns of sensitivity? We propose to answer these questions by applying the accumulated knowledge gained through numerous watershed analyses in the project area, plus existing theoretical and empirical information. From this information, we have developed a system that guides riparian management activities on stream reaches throughout the drainage network, and in different geographic locations (e.g., western Washington vs. the Interior Columbia River Basin). The process we developed leads to pre-determined management actions for readily identified situations (Figure 2). Refinement is made possible through more intensive analyses (i.e., watershed analysis, stream guilding, monitoring, and research). This approach to problem solving is not unlike that used for the stream guilding process described by Watson et al. (1998a). In this instance, it is applied specifically to riparian areas.

The riparian management strategy developed herein focuses on the large woody debris (LWD)-providing function of riparian forests. This assumes that the greatest constraints on riparian timber management will, in most cases, be imposed by the need to provide LWD. Other important functions (bank integrity, shade, leaf litter, etc.) are assumed to be adequately addressed by LWD prescriptions. Evidence from other studies and watershed analysis suggest that this is reasonable (Figure 3).

Figure 3. Generalized curves depicting riparian forest effects on streams as a function of distance from the channel (from USDA et al. 1993).



The purpose of this report is to provide the technical foundation to formulate a riparian management strategy like "C" in Figure 1, for use in Plum Creek Timber Company's Native Fish HCP. We describe the conceptual framework, scientific principles, and analyses that underpin this strategy. Management prescriptions, per se, are not developed in this document. These will be developed in concert with other measures, such as those for stream temperature, so as to allow for evaluation of operational and economic implications. However, gaps in existing forest practice protection measures, if any, are identified through an evaluation process. Relevant principles of riparian structure, function, and anticipated response to timber harvest are provided to managers to address any identified gaps. Future trends at the stand and watershed level are also addressed.

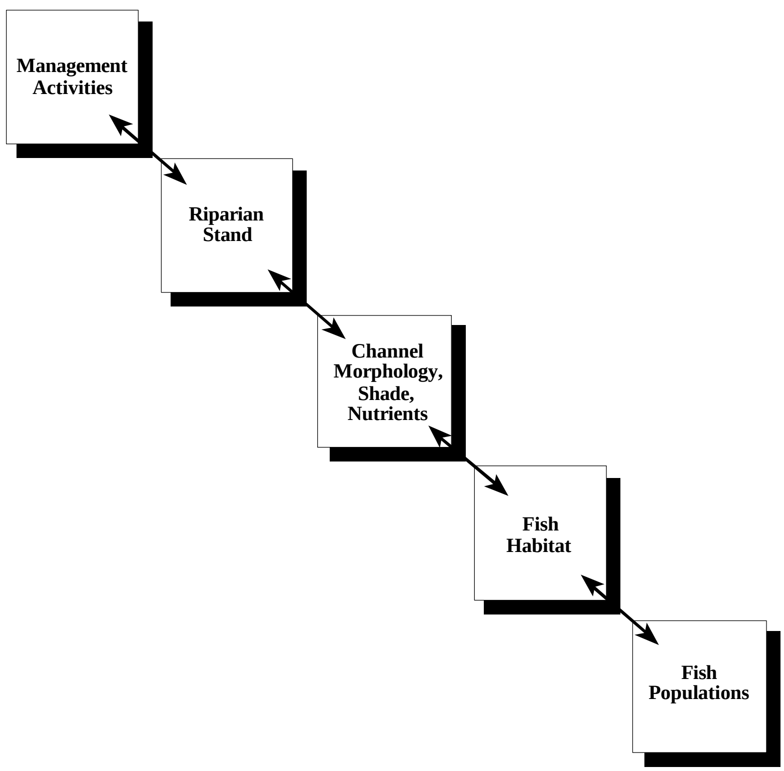
Specific objectives of this assessment are to (1) describe differences in fish habitat sensitivities within a drainage network, (2) develop a process for quantitatively evaluating outcomes of various management scenarios, (3) apply this tool to evaluate existing management prescriptions, and (4) identify any gaps in existing practices (i.e., the steps in Figure 2).

2.0 BACKGROUND AND CONCEPTUAL FRAMEWORK

Not all stream reaches in a drainage provide equal types and quantities of fish habitat. Streams vary in their inherent capacity to support a particular type of habitat, and fish species or life history stages occupy only specific reaches at a particular time. Identification of these patterns of habitat type and use provides a template for tailoring management objectives and actions to properly address stream ecosystem needs. To identify and incorporate patterns in fish use and channel sensitivities, we employed the concept of the Riparian Cause-Effect Pathway.

2.1 The Riparian Cause-Effect Pathway

Figure 4. The Riparian Cause-Effect Pathway



The connections between management activities and fish populations can be described in a cause-effect fashion (Figure 4) (Washington Forest Practices Board 1995, Schuett-Hames et al. 1998). Ideally, the workings of this pathway would be so well understood that all actions would have immediate and predictable outcomes in time and space. An improperly installed road culvert that blocks access to migrating fish is an example of this type of direct cause-effect linkage. More often, the linkages are less direct and poorly understood. An example of an indirect fish-riparian relationship is provided by Hartman (1988), who linked riparian timber harvest to changes in fish production through influence on the annual temperature regime of Carnation Creek, B.C. Rarely do we have such a firm understanding of cause-effect relationships along the riparian pathway.

Changes in riparian input processes are generally expected to have some influence on fish populations. Large woody debris (LWD) contributed from riparian forests can modify local channel morphology, which in turn can affect availability of spawning and rearing habitat for salmonids, which ultimately can affect populations. However, not all riparian areas contribute the same types and quantities of LWD[8]; not all channels respond in the same manner to LWD (Montgomery and Buffington 1997); and different channels support different amounts and quality of fish habitat (see discussion below). It is the recognition and management of these differences in channel sensitivities and fish habitat vulnerabilities that are the foundation of Plum Creek Timber Company's approach to riparian management.

In this section, the role of LWD in streams will be examined from two perspectives; channel morphology and fish habitat. Patterns in these features will then be used in evaluation of existing and potential riparian management strategies.

2.2 The Role of Large Woody Debris in Streams

In forested riparian areas of the Pacific Northwest, the presence of LWD[9] in streams is considered a significant component of stream habitat for fish (Harmon et al. 1986; Bisson et al. 1987; Sedell et al. 1988; Bjornn and Reiser 1991). LWD physically alters stream flow patterns and channel characteristics, provides cover, and represents a long-term food source for aquatic organisms. LWD dissipates stream energy and creates local channel scour and deposition.

Although there is general consensus that LWD contributes to diversity within stream channels, which is beneficial to fish and other aquatic organisms, there is less agreement on the amount of LWD that is necessary to support viable fish populations. To assume that fish populations are strongly correlated with LWD implies that fish populations in forested areas are limited by the amount of LWD in all streams. This may or may not be true depending on stream size and gradient, valley morphology, and other factors. For example, the relationship between LWD and pool area or pool spacing has been shown to vary with channel slope and channel width in western Washington streams (Bilby and Ward 1989; Beechie and Sibley 1997, Montgomery and Buffington 1997) (see discussion in the "channel

[8] Some stream reaches flow through riparian areas where LWD contributions are not possible, e.g., meadows.

[9] There is no established standard for LWD dimensions, but most researchers report a minimum diameter of 10 cm (~4 in), and a minimum length of 1- 2 m (~3.3 - 6.6 ft).

classification" section below). Economic implications of retaining merchantable trees for LWD require a high degree of confidence that the LWD is necessary and will meet the needs of native fish.

In this section, we briefly explore some aspects of the ecology of LWD in streams and rivers. We focus mostly on the riparian pathway elements that concern the influence of LWD on channel characteristics, fish habitat, and fish populations.

2.2.1 Influences of LWD on Streams and Rivers

In many streams, LWD is important in controlling formation of aquatic habitat and the movement of sediment and water. Physical processes associated with LWD in streams include the formation of pools and other important rearing and spawning areas, control of sediment and organic matter storage, and modification of water quality (Harmon et al. 1986). In general, the influence of LWD on stream ecosystems is thought to decrease in a downstream direction (Vannote et al. 1980).

2.2.1.1 Channel Morphology

LWD increases the hydraulic roughness of a channel and tends to locally influence the time-rate dissipation of potential energy of flowing water. Roughness also depends on other factors such as particle sizes of bed and bank materials, stream sinuosity, bank characteristics, and streamside vegetation. Features such as LWD are particularly important for low gradient, unconstrained streams because they dissipate energy and stabilize channels (Bisson et al. 1987).

LWD has been shown to be important in the formation, abundance, and geometry of pools and riffles. Bisson et al. (1987) describe in detail how pools are formed or their geometry is modified by scour and deposition associated with stream flow over, under, and around LWD (see discussion under The Role of LWD Relative to Channel Class). They also describe how LWD can increase pool frequency and variability in pool depths. Grette (1985) reported a significant correlation between pool frequency and debris pieces in low gradient western Washington streams. Hilderbrand et al. (1997) found that pool area increased 146% in sites with systematic placement of LWD and 32% in random-placement sections of a low-gradient stream. In addition, pool frequency and volume decreases in stream channels when LWD is removed (Bisson and Sedell 1984). Bilby (1984) also reported reduction in number and area of pools after removing LWD from a 11.5-m-wide segment of Salmon Creek, Washington.

Longitudinal profiles of streams can be influenced by LWD. Steps in the longitudinal profile are created where a large log or accumulations of LWD form a dam that traps a wedge of sediment (Bisson et al. 1987). The proportion of the channel gradient regulated by LWD ranges from about 10-100% in first- to fifth-order streams; the effects of LWD on channel profile decrease with increasing stream order (Harmon et al. 1986). For example, Bilby (1981) reported that channel drop formed by LWD decreased from 52% to 46% to 10% from small first-order streams to larger third-order streams in New Hampshire. Marston (1982) evaluated the effects of log steps in 13 watersheds of up to fifth order in the Oregon Coast Range. He found that LWD steps controlled only 6% of total fall in those channels, and suggested that geologic factors, not LWD, controlled the overall shape of the longitudinal profile. The degree to which LWD controls stream profile is related to abundance and size of LWD and the ability of channels to bypass obstructions (Harmon et al. 1986).

Because LWD dissipates stream energy by obstructing stream flow, it increases the complexity of channel habitats. According to Bisson et al. (1987), LWD maintains stream habitat diversity by anchoring the position of pools along the thalweg (deepest portion of a stream channel), creating backwater habitat along the stream margin, causing lateral migration of the channel and the formation of secondary channel systems in alluvial valley floors, and by increasing water depth variability.

Although most LWD studies were conducted in coastal forests of the Pacific Northwest, the role of LWD in Rocky Mountain streams appears to be quite similar. Richmond and Fausch (1995) measured LWD in 11 undisturbed streams draining subalpine old-growth forest in the Rocky Mountains to assess abundance, characteristics, and function. They found that although LWD in Rocky Mountain streams had smaller diameter, length, and volume than in coastal Pacific Northwest streams, its abundance and function were similar. They noted that the majority of pools (76%) were plunge and dammed pools formed by LWD and that smaller streams had a greater proportion of pool-forming pieces than larger streams. Richmond and Fausch (1995) concluded that the function of LWD in forming fish habitat in small Rocky Mountain streams was strongly influenced by the stream's location within the watershed. Stream size and gradient appeared to exert significant influence on the characteristics and function of LWD in the studied streams.

2.2.1.2 Sediment and Organic Matter Storage

LWD creates a temporary storage of inorganic sediment and organic matter in stream channels. Megahan (1982) found that LWD accounted for 35% of the obstructions and 49% of the stored sediment volume in seven first- or second-order watersheds in Idaho. Total sediment stored in these watersheds was roughly 15 times the mean annual sediment yield. Further evidence for the role of LWD in sediment storage comes from LWD removal in which sediment and organic matter transport rates increase several-fold after removal of LWD. For example, Bilby (1981) compared sediment yields from a 175-m section of stream in a second-order New Hampshire watershed before and after removal of LWD. In the first year after removal, export of fine and coarse particulate matter increased 500% over the value expected for the untreated condition. Export of dissolved organic matter in this study also increased with LWD removal, but only by 6%. MacDonald and Keller (1983) observed that 60% of the sediment stored in a 100-m stream segment was transported out of the reach during the first winter after LWD removal.

Speaker et al. (1984) found that stream reaches with LWD retained leaves roughly 10 times more efficiently than reaches without LWD accumulations. These researchers noted that LWD trapped branches and sticks that in turn trapped leaves. Roots, stems, and other woody debris can play an important role in a stream's nutrient flux by trapping carcasses of post-spawning adults in streams inhabited by anadromous salmonids. The importance of the nutrients derived from these carcasses is only beginning to be realized (Cederholm et al. 1989, Bilby et al. 1996, Levy 1997). Thus, the stability and storage capacity of LWD is enhanced by the presence of branches and roots, which help to anchor the debris and serve as a matrix to trap and consolidate sediment, fine particulate organic matter, and coarse particulate organic matter in the form of fish carcasses.

2.2.2 Influences of LWD on Fish

Because LWD potentially provides cover, creates important hydrologic features such as pools and backwaters, and stores sediments, it is considered an important component of fish habitat (see reviews given by Bryant 1983; Sedell et al. 1985; Harmon et al. 1986; Bisson et al. 1987; Maser and Sedell 1994). The relationships between LWD, fish populations, and their habitat are described in more detail below.

One of the more important functions of LWD in forming fish habitat is the creation of rearing pools (Bisson et al. 1987). Salmonids select stream locations predominantly on the basis of velocities where net energy gain (potential profit) and hence fitness are maximized (Fausch 1984). That is, salmonids occupy pockets of slow water near high velocity, food-bearing waters. LWD tends to break swift flows into pockets of quiet water (e.g., pools and backwater areas), which salmonids often use. However, Wilzbach (1985) and Wilzbach and Hall (1985) found that habitat complexity, including the structural complexity created by LWD, may cause a reduction of foraging efficiency by obscuring food items from cutthroat trout. These researchers posited that the benefits gained from increased cover may be partly or completely offset by reduced food consumption, especially when prey abundance is low. This may be why Spalding et al. (1995) found that juvenile coho were not attracted to brushy debris and that brushy debris did not enhance coho survival or growth. Bisson et al. (1987) note that LWD offers salmonids cover from predators, high flows, and winter icing conditions; however, the degree of protection provided by LWD is largely unquantified.

Sediments deposited upstream from LWD accumulations serve several important functions for fish populations, including food production sites, sorting of spawning gravels, and retention of fine sediments. Salmonids use the spawning gravels that form both upstream and downstream from LWD accumulations (Sedell et al. 1982). Both the LWD itself and the organic material it stores are considered an important energy source for stream invertebrates. Smock et al. (1989) found that increasing the abundance of LWD dams increased organic matter storage, retention of leaves, and invertebrate abundance. These detritus-based invertebrates are a principal food for salmonids (Mundie 1974). The other important benefit of sediment storage is the moderating effect of LWD on sediment transport rates. LWD buffers the channel against rapid changes in sediment loading that could degrade spawning gravels, fill rearing pools, and reduce invertebrate production (Bisson et al. 1987).

Because of the habitat complexity formed by LWD in streams, salmonids are frequently found associated with these structures. For example, Shirvell (1990) found that juvenile coho and steelhead occupied previously infrequently-used stream areas after rootwads were placed near mid-channel. Nearly all coho and 83% of steelhead occupied positions associated with rootwads during drought, normal, and flood streamflows. Positive correlations between coho numbers and LWD have been observed in Oregon streams (House and Boehne 1986) and Alaskan streams (Murphy et al. 1985).

Bisson et al. (1982) noted that juvenile steelhead and cutthroat of all ages preferred habitats with abundant LWD in Washington streams.

Although LWD has an important influence on channel characteristics of streams and fish appear to be attracted to these areas, does the presence of LWD result in a greater abundance of fish? One way to test this is to experimentally manipulate the amount of LWD in streams and concomitantly monitor the response of fish populations. Several studies have examined the effects of LWD removal on fish populations. For example, Elliott (1986) assessed the effects of removing LWD on Dolly Varden in southeastern Alaska. He found a progressive loss of larger fish and a decreased mean size of individuals after debris removal. Dolloff (1986) observed declines in both coho salmon and Dolly Varden in two, second-order streams in Alaska after small or unstable pieces of debris were removed. Bryant (1982) also reported a decline in numbers of juvenile coho salmon after removal of LWD in two Alaska streams. Lestelle and Cederholm (1984) found that removing LWD from two Washington streams resulted in a short-term decline in resident cutthroat trout numbers. Within one year, however, the numbers of cutthroat returned to pretreatment levels apparently because of rapid recruitment of LWD. Fausch and Northcote (1992) compared salmonid populations in sections of a small British Columbia stream that had been cleaned of LWD with sections where debris was relatively undisturbed. Three sections where debris had been removed had simple habitat that was less sinuous, wider, and shallower and had less pool volume and overhead cover than the sections with more complex habitat where debris was retained. Fausch and Northcote (1992) found that biomass (kg/ha) and individual weights of coho and cutthroat were significantly greater in complex than in simple sections. They estimated that five times the current biomass was forgone because of debris removal.

Another way to test the relationship between LWD and fish abundance is to record the response of fish populations to additions of LWD. In a small coastal stream in Washington, Cederholm et al. (1997) compared the response of juvenile coho and steelhead to addition of LWD in two treatment sections with an unaltered (reference) section. The abundance of LWD was 7.9 times greater than the pretreatment level in one treatment section and 2.7 times greater in the other; abundance of LWD remained unchanged in the reference section. After addition of LWD, winter populations of juvenile coho and smolt yields increased significantly in both treatment sections, but decreased slightly in the reference site. Populations of juvenile steelhead showed no response to additions of LWD. In some studies the response of salmonids to additions of LWD have been mixed (see Beschta et al. 1991; Platts 1992). Perhaps in some cases LWD was not the limiting factor. Chapman (1995) reviewed a number of these studies and concluded that several factors, including insensitive experimental designs, invalid monitoring designs, and lack of full seeding confounded these studies.

The studies described above indicate that a decreased supply of LWD can change the quality of salmonid habitat by reducing cover, pool habitat, protection from high flows, storage of gravels and organic matter, and hydraulic complexity. The potential consequences for salmonid growth and survival include increased vulnerability to predation, lower winter survival, reduced rearing and spawning habitat, reduced food production, and loss of species diversity (Hicks et al. 1991). However, this does not address the question of how much LWD is too little or how much is too much. Given that the habitat requirements of various salmonid species, as well as various age groups, often differ, there is no simple answer. The problem is further complicated because manipulation of LWD in natural streams has in some cases given equivocal results. Bisson et al. (1987) suggest that actual estimates of the quantities of LWD needed for adequate fish habitat are best obtained from pre- and post-logging assessments.

Salmonid populations evolved in environments characteristic of unmanaged streams. Depending on the characteristics of the stream and its basin (see discussion below), LWD may or may not have been an important component of the stream environment. Many streams in the Pacific Northwest probably offered debris-rich environments, and the species developed adaptations that enabled them to maximize production in hydraulically complex channels. It is important, therefore, to know natural LWD levels and loading rates in various streams. Knowledge of debris loadings typical of streams of similar size in the same geographical area or of the same geomorphic guild (see Watson et al. 1998[a]) will help to provide references against which changes caused by management activities can be evaluated (Bisson et al. 1987).

2.3 Channel Sensitivity to LWD Based on Slope Class

2.3.1 Fish Use as a Function of Channel Slope

Distribution and abundance of native salmonids are important considerations when assigning relative LWD sensitivities to different portions of the channel network (Nelson et al. 1992, Reiman and McIntyre 1995). Plum Creek Timber Company has completed extensive presence / absence surveys for fish throughout the project area. These surveys demonstrate that fish are highly concentrated in lower gradient reaches (Table 1). For example, of all the reaches that contained bull trout, 88% had gradients <7%, and about 79% had gradients <3%. At least 96% of the reaches that contained salmonids had gradients <10%. These observations comport with those of Kruse et al. (1997), who found similar relationships between channel slope and cutthroat trout distribution.

Table 1. Cumulative percent of fish use by channel gradient class. *The gradient class "0%" includes channels with measured gradients £ 1%, the gradient class "1%" includes channels with gradients of 1 to 2%, and so on. (Plum Creek Timber Company, unpublished data).*

Gradient Class (%)	Cumulative Percentage Distribution of Observed Fish						
	Bull Trout	Brook Trout	Cutthroat Trout	Whitefish	Coho Salmon	Brown Trout	Rainbow Trout
0	1.7	4.5	0.9	3.3	0.0	2.8	2.4
1	36.4	41.5	21.9	40.0	7.1	27.8	32.2
2	62.0	70.2	50.0	80.0	53.6	66.7	62.9
3	78.9	81.8	69.9	100.0	75.0	77.8	80.4
4	82.6	88.6	80.3	100.0	85.7	88.9	90.2
5	85.1	92.4	86.5	100.0	89.3	100.0	93.7
6	88.0	95.3	91.4	100.0	100.0	100.0	96.5
7	90.5	97.7	93.4	100.0	100.0	100.0	99.0
8	92.6	98.6	95.0	100.0	100.0	100.0	99.3
9	95.0	99.0	96.0	100.0	100.0	100.0	99.7
10	96.7	99.2	96.9	100.0	100.0	100.0	99.7

Table 2. Channel morphologic classes and corresponding typical channel gradient classes defined by Montgomery and Buffington (1997). *The gradients are measured at the reach scale (10^2 to 10^3 ft). Note that approximately 90% of bull trout habitat (defined by bull trout presence) is found in three morphologic types: pool-riffle, plane-bed and step-pool.*

Channel Morphology	Typical Gradient Range (%)
Pool-riffle	gradient ≤ 1.5
Plane-bed/Forced pool-riffle	1.5 < gradient ≤ 3
Step-pool	3 < gradient ≤ 6.5
Cascade	6.5 < gradient

2.3.2 Stream Channel Classification

A geomorphic basis for classifying stream channels according to morphology and potential channel response is described by Montgomery and Buffington (1993, 1997), based on work done in Washington and Alaska. Channel gradient is the primary control on channel morphology in this classification system (Table 2), although additional aspects of fluvial processes and corresponding morphology are used to segregate channel types. These gradient classes are intended to represent typical relationships between gradient and channel morphology; some degree of overlap between classes in Table 2 is implicit. In field applications, channels frequently exhibit more than one of the morphologic types, hence the classifications are typically a blend of morphologic types existing in a continuum (Montgomery and Buffington 1997). Brief descriptions of the primary distinguishing features of the morphologic types are provided below (excerpted from Montgomery and Buffington 1997).

Cascade channels are streams

> *"…in which energy dissipation is dominated by continuous tumbling and jet-and-wake flow over and around individual clasts (e.g., boulders, cobbles). Cascade channels generally occur on steep gradients, are narrowly confined by valley walls, and are characterized by longitudinally and laterally disorganized bed material consisting of cobbles and boulders." (p. 597).*

> *"Step-pool channels are characterized by longitudinal steps formed by large clasts organized into discreet channel-spanning accumulations that separate pools containing finer material." "Step-pool morphology is associated with steep gradients, small width to depth ratios, and pronounced confinement by valley walls." (p. 597)*

> *Plane-bed channels "…encompass glide (run), riffle, and rapid morphologies described in the fisheries literature. Plane bed channels lack discrete bars…[and] occur at moderate to high gradients in relatively straight channels that may be either unconfined or confined by valley walls." "They are…dominantly gravel to cobble bedded. Plane-bed channels differ…from both step-pool and pool-riffle channels in that they lack rhythmic bedforms and are characterized by long stretches of relatively featureless bed." "Plane-bed channels typically exhibit armored bed surfaces calculated to have a near-bankfull threshold for mobility, although elevated sediment loading can cause textural fining…" (p. 600)*

> *"Pool-riffle channels have an undulating bed that defines a sequence of bars, pools and riffles. This lateral bedform oscillation distinguishes pool-riffle channels from the other channel types…" "Pool-riffle channels occur at moderate to low gradients and are generally unconfined, and have well established flood plains. Substrate…varies…but typically is gravel sized."*

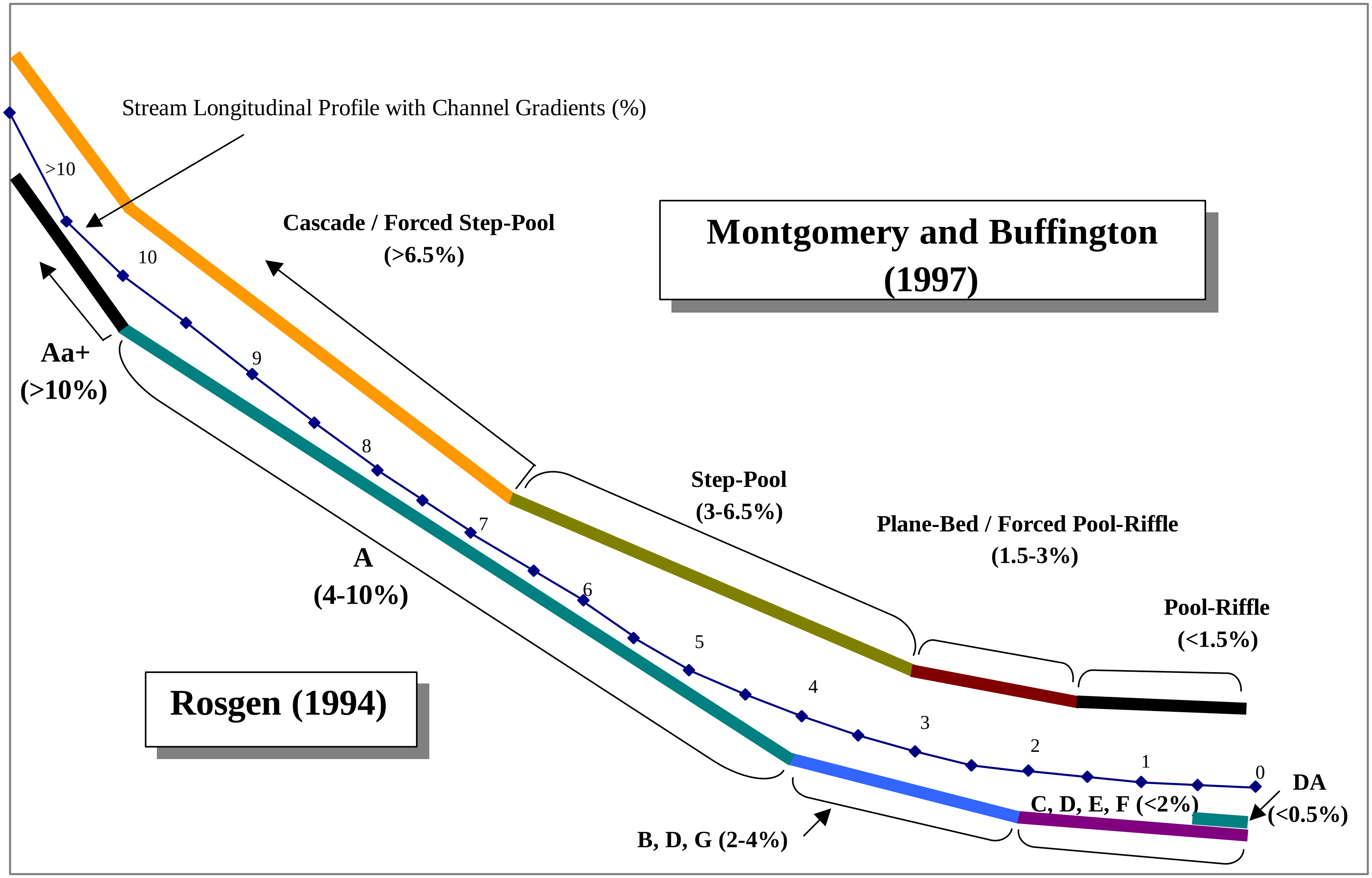
In addition to the main channel types described above, forced morphologies may occur when roughness elements such as large woody debris are introduced to a channel; this may accompany deviation from the typical gradient range for a given morphology. The most common forced morphologies are forced pool-riffle and forced step-pool (Montgomery and Buffington 1997). Forced morphology results from bedrock outcrops (or other erosion-resistant material such as cemented or cohesive volcanic or glacial/lacustrine deposits) in the channel bed or banks, or from abundant LWD.

2.3.2.1 The Rosgen Classification System

The stream channel classification system developed by Montgomery and Buffington (1993) is not the only one of its kind. A similar system developed by Rosgen (1994) is widely used throughout the western United States.

Like Montgomery and Buffington, channel gradient is a fundamental classification parameter in the Rosgen system. However, unlike Montgomery and Buffington, Rosgen goes further to describe channel sub-types using streambed material size classes. Both systems offer insight into potential channel responses to changes in watershed inputs (e.g., wood, water, and sediment). Recent investigations into watershed structure and function within the Planning Area and elsewhere in the Pacific Northwest via Watershed Analysis (WFPB 1995) have employed the Montgomery and Buffington system, and therefore it is this system that we have selected for use in this report. For the convenience of readers who are more familiar with the Rosgen system, we compare the two systems on the basis of stream gradient in Figure 5.

Figure 5. Longitudinal Profile of a Stream Channel and Stream Gradient Classes Used to Distinguish Channel Types in Rosgen's (1994) and Montgomery and Buffington's (1997) Channel Classification Systems.



2.3.3 The Role of LWD Relative to Channel Class

Channel classification work by Montgomery and Buffington (1997) has promoted better understanding of how LWD interacts with channel classes to form fish habitat (e.g., pools). Montgomery et al. (1995) investigated pool frequency as a function of LWD abundance in the context of plane-bed channels (approximately 1.5% - 3% gradient). They found that the normally low frequency of pools in plane-bed channels could be significantly increased through addition of LWD. In the absence of LWD (or other flow obstructions), channels in the plane-bed gradient class would be expected to have few pools and relatively uniform beds composed of gravel, cobbles and boulders. Consequently, channels in the gradient class for plane-bed morphology are expected to be highly sensitive to LWD load, particularly with respect to pool frequency.

Pool-riffle reaches (nominal gradient < 1.5%, Montgomery and Buffington 1997), are characterized by rhythmic stream meander patterns, and the curvature of the bend induces cross-stream circulation that scours pools and forms riffles (e.g., Dietrich and Smith 1983). Bank erosion at the outside of the bend is also characteristic of this channel type. The processes that characterize pool-riffle morphology dictate a pool frequency that is typically about 5 to 7 channel widths (e.g., Leopold et al. 1964). Consequently, it can be concluded that pool-riffle channels will have a significant, inherent pool frequency that is largely unaffected by LWD abundance, and are therefore less sensitive to LWD supply than plane-bed/forced pool-riffle channels. Smith et al. (1993) demonstrated this in their study of the effects of LWD removal on channel morphology in a 1% gradient stream in southeast Alaska. No changes in mean pool spacing or residual pool depths were detected within four year of LWD removal.

Montgomery et al. (1995) found that step-pool morphology is relatively insensitive to LWD abundance. Pool frequency was not related to LWD abundance. Nevertheless, about 40% of pools in step-pool systems were interpreted to have been created by LWD. In addition, Montgomery and Buffington (1997) acknowledge that forced step-pool morphology can be caused by LWD. Observations of step-pool streams in Washington and Montana (e.g., O'Connor 1997a, b) indicate that LWD steps are significant components of channel morphology in some channels in the step-pool gradient class. Consequently, step-pool channels are judged to be less sensitive to LWD than plane-bed/forced pool-riffle channels, but likely to respond morphologically to LWD inputs.

As with pool-riffle channels, a significant number of pools can be expected in step-pool channels independent of LWD load, but additional LWD can be expected to increase pool frequency in many cases (i.e., forced step-pool). In some cases, LWD may be the critical element of channel morphology. In the context of Montgomery and Buffington's channel classification system, the loss of LWD recruitment would tend to shift forced pool-riffle and forced step-pool reaches toward plane bed morphology. Plane-bed morphology is relatively poor-quality habitat for most salmonids owing to a lack of diversity of water depth and velocity as well as a lack of cover.

2.3.3.1 Relationships Between LWD Frequency, Pool Spacing, and Channel Gradient in Montana

The hypotheses of channel response to LWD presented in Montgomery et al. (1995) and amplified in the discussion above have been tested on two existing data sets from Montana watersheds, the Swan and Thompson rivers. Both of these drainages lie within the project area. Data for pool spacing (pools per channel width) were regressed as the dependent variable against LWD frequency (LWD pieces per meter of channel length) for three gradient classes (1%, 2-3%, and 4-6% roughly representing pool-riffle, plane-bed/forced pool-riffle, and step-pool morphologies, respectively). Summary statistics for these regression equations for each of the six cases are shown in Table 3.

The regression analysis revealed that LWD does significantly influence pool spacing in both watersheds in the plane-bed/forced pool-riffle channels (represented by the 2-3% gradient class). In pool-riffle channels (1% gradient class), there was no influence on pool spacing. In step-pool channels (4-6%), there was a significant relationship in the Swan River basin, but not in the Thompson River basin. These data support the overall assessment of channel sensitivity to LWD for the project area, as summarized in Table 4. Streams with gradients of 6% or less are presumed to be most sensitive to LWD. These channels also represent the bulk of the drainage networks that are occupied by fish (Table 1). Consequently, the greatest sensitivity to LWD is assigned to channels with gradients of 1.5-3% owing to the demonstrated influence of LWD on pool spacing.

Table 3. Regression equations and statistical significance for pool spacing as a function of LWD frequency in two Montana watersheds; * denotes significance @ 95% confidence level. *Regression equations were of the form $y = ax^b$, where y is pool spacing and x is LWD frequency.*

Watershed and Gradient Class	*a*	*b*	r^2	p
Thompson 1%	8.58	-0.594	0.13	0.38
Thompson 2-3%	6.35	-0.264	0.11	0.02*
Thompson 4-6%	6.19	-0.221	0.04	0.39
Swan 1%	4.01	-0.294	0.15	0.14
Swan 2-3%	0.98	-1.27	0.76	<0.00001*
Swan 4-6%	2.65	-0.76	0.52	0.01*

2.3.3.2 Channel Migration Zones

In addition to the relationships between channel morphology and sensitivity to LWD discussed above, a channel's propensity to migrate laterally across its valley bottom may influence channel sensitivity to LWD. Certain stream reaches in river systems, so-called "channel migration zones" (Washington Forest Practices Board 1995), may be observed to have relatively high rates of lateral channel migration. The recognition of these types of channels has raised concerns that application of regulatory riparian buffers along channel migration zones may fail to account for future channel movement. O'Connor and Watson (1998; Appendix A) prepared a system of classification to identify channel migration zones (CMZs) on Plum Creek lands in the project area.

The classification system includes five CMZ types, all of which include secondary or multiple channels on the floodplain as a diagnostic characteristic. CMZ types are distinguished on the basis of channel entrenchment (confinement), channel gradient, migration process(es), and spatial extent and/or continuity of CMZs, among other factors. The system also hypothesizes relative channel migration potential of each CMZ type, as well as sensitivity of each to inputs of coarse sediment and LWD.

The general mechanism of channel migration proposed by O'Connor and Watson (1998) is that excess coarse sediment and/or LWD inputs to a channel have the potential to reduce the channel's capacity to convey streamflow to such an extent that a given flow may be forced out of the channel

Table 4. Summary of sensitivity to LWD by channel type. *The gradient for the upper bound of the step-pool morphology class has been modified for convenience to 6% from 6.5% (Montgomery and Buffington 1997). This adjustment is consistent with the concept of a continuum of channel morphologies across gradients* **.**

Channel Type (Gradient Class)	LWD Sensitivity	Rationale
Pool-riffle (< 1.5%)	Moderate	Pool-forming processes are significant in the absence of LWD, however, there is evidence that LWD can increase pool frequency and provide other significant habitat elements.
Plane-bed / Forced pool-riffle (1.5%-3%)	High	Pool-formation is significantly affected by LWD; loss of LWD from forced pool-riffle reaches may cause shift to plane-bed morphology. Recruitment of LWD to plane-bed channels introduces forced pool-riffle morphology.
Step-pool (3%-6%)	Moderate	Pool-forming processes are significant in the absence of LWD, however, there is evidence that LWD can increase pool frequency and provide other significant habitat elements.
Cascade (>6%)	Low	Pool-forming processes are significant in the absence of LWD, although there is evidence that LWD can increase pool frequency and provide other significant habitat elements.
Channel Migration Zones	Moderate or High	Where CMZs are identified, the sensitivity rating for the CMZ type takes precedence over the gradient-based rating. This is most significant with respect to channels in the step-pool gradient class where a moderate rating may be replaced by a high sensitivity rating.

and on to the floodplain (avulsion). If floodplain topography and gradient are sufficient to concentrate flow, a new channel may be cut, potentially at any location in the floodplain. This is thought to be the primary mechanism in channels with steeper gradients (plane-bed, forced pool-riffle and step-pool channels).

In channels with relatively low gradient, such as pool-riffle channels, gradual bank erosion is more likely to be the dominant mode of channel migration, although avulsion may also occur (e.g., meander bend cut-off channels). The main distinction here is that bank erosion processes are incremental, and are not as likely to result in a new channel at an unpredictable location on the floodplain as in the case of channel migration by avulsion. This distinction is relevant to riparian forest management in areas with relatively high rates of channel migration. Different management strategies may be required to maintain long-term riparian forest contributions to stream ecosystem function (e.g., wider riparian management zones, no harvest buffer zones).

In any case, channel migration on a forested floodplain would recruit LWD from the riparian stand. Such relatively intense LWD recruitment would likely create positive feedback[10] in that in-channel LWD would increase bank erosion potential as well as increase channel roughness, thus reducing available energy to transport sediment. At the same time, bank erosion or avulsion would add coarse sediment to the stream, further increasing potential for channel migration. Thus in some CMZs, it is likely that LWD plays a critical role, and long-term reduction in LWD recruitment to these stream reaches could fundamentally alter local channel morphology.

Where CMZs are identified, particularly in CMZs where forced step-pool morphology is typical (e.g., Type C; Appendix A), LWD is assumed to play a critical role in channel morphology. In this circumstance, LWD sensitivity would be considered high.

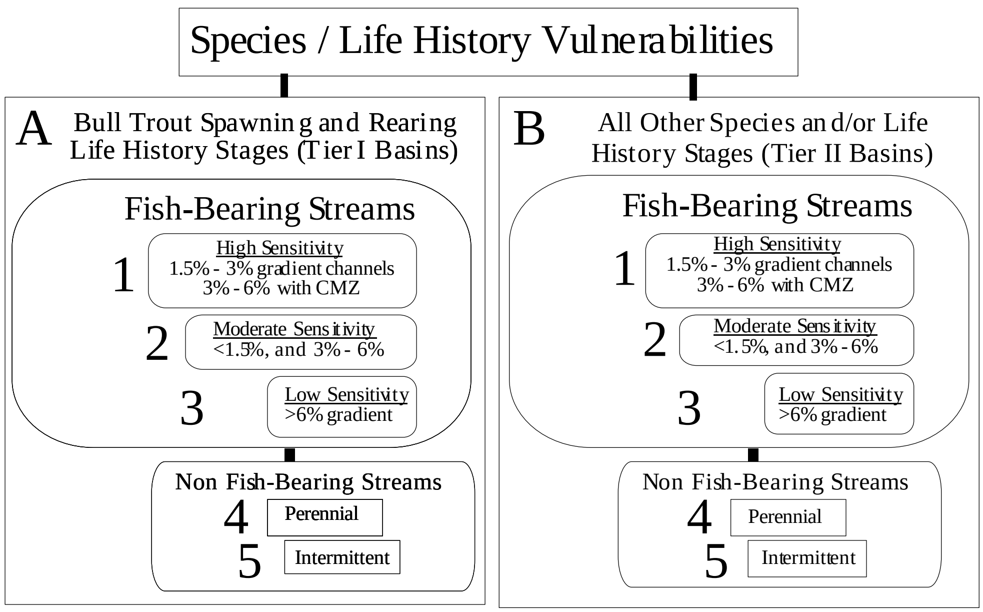
The patterns of channel morphology and sensitivity to LWD described above, including channel migration zones, have been observed repeatedly during watershed analyses in Washington, Oregon, and Montana. Channel sensitivities presented in Table 4, and their associated fish habitat vulnerabilities, provide a means of assigning priority to riparian protection.

[10]This assumes channel width and depth are not so large that LWD is routed downstream.

2.4 Prioritization of Riparian Protection Levels

Information on fish habitat vulnerabilities and channel sensitivities derived from published studies and field research, as previously discussed, were used to prioritize riparian protection levels (Figure 6). Separation by species and life history stage follows the definition of "Tier I" and "TierII" watersheds[11], as defined by the Plum Creek Native Fish HCP, and reflects the emphasis on bull trout in the project area. All separations, i.e., by channel class or life history stage, are based on best available information regarding cause-effect relationships, including field surveys in the project area.

Figure 6. Prioritization of riparian protection levels for the Plum Creek Timber Company Native Fish HCP*.



*This prioritization applies to the LWD recruitment function of riparian forests. See text for discussion of shade and other riparian functions.

[11] Tier I watersheds contain stream reaches that support spawning and juvenile rearing life stages of bull trout. Tier II basins contain stream reaches that support adult migration, foraging, and winter rearing life stages of bull trout, and all life stages of other native salmonids (e.g., westslope cutthroat).

Fish-bearing streams are assigned highest priority in this fashion. The cause-effect linkages between forest management activities and resource effects are best understood in this portion of the drainage network. Timber harvest will have the most direct and immediate effect on associated stream conditions in fish-bearing reaches. Fish-bearing streams constitute an estimated 827 miles in the project area (i.e., on Plum Creek lands), based on GIS-calculated stream miles. Protection measures instituted at this level have the highest probability of success. Perennial, non fish-bearing waters are assigned somewhat lesser, but appropriate protection. These streams are considered important primarily as conduits of watershed inputs such as sediment and heat energy to fish-bearing waters. Cause-effect linkages for LWD are less well understood in this portion of the drainage network, and response to actions may be indirect, or delayed. Streams of this type constitute an estimated 1,051 miles in the project area. Protective measures instituted at this level have a lower or less tangible probability of benefiting fish populations compared with the economic cost of management restrictions.

Intermittent channels receive the lowest relative level of protection. Their role is also seen primarily as conduits for watershed inputs (sediment, nutrients, water, and on rare occasion, heat energy). These channels are sensitive to forest management activities that deliver sediment (e.g., soil disturbing activities, mass wasting, road sediment, etc.) that can be transmitted during episodes of connection with the perennial drainage network. Therefore, attention to sediment inputs is of paramount importance in this portion of the drainage network. LWD can help store sediment in these channels (Commandeur et al., 1996). Intermittent channels have little to do with stream heating and cooling, because when these channels typically flow in the spring or early summer, stream temperatures in fish-bearing reaches are not at their annual extremes (see Sugden et al. 1998). Intermittent channels constitute an estimated 3,185 miles in the project area.

The intent of prioritizing protection levels is to convey to managers the relative sensitivity, and therefore relative importance, of different combinations of channel class and fish species. By responding to riparian situations according to the priorities exhibited in Figure 6, timber management is expected to ensure continued maintenance and survival of native fish stocks in the project area. All portions of the drainage network are encompassed by the stratification, as all stream reaches warrant some consideration during timber management activities. The "low" category does not imply "no" sensitivity; simply that with respect to LWD, these reaches are relatively insensitive to gain or loss of LWD[12], and management can therefore emphasize economic objectives without high risk to fish resources. Conversely, the "high" category does not imply these reaches are so sensitive to LWD gain or loss that every piece of LWD is crucial to maintaining viable fish populations. It does imply that these reaches, above all others, should be managed with fish habitat as the principal objective. This approach is one of "scaled sensitivity", considering: (1) magnitude of impact, (2) likelihood of success, and (3) economic cost. The intent is to match management attention to ecological sensitivity in a cost-effective manner.

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[12] In stream reaches where LWD is not limiting, other functions such as canopy closure for maintenance of stream temperatures may take precedence.

Table 5. Conceptual array of riparian management options (prescriptions) designed to address in-channel large woody debris (LWD) loads for combinations of channel sensitivity and fish species / life history stage habitat vulnerability. *The management options would apply to fish-bearing stream reaches on Plum Creek Timber Company ownership in the project area.*

Fish Species / Life History Stage Vulnerabilities	Channel Sensitivity by Stream Gradient Class[1]		
	High	**Moderate**	**Low**
	1.5% < gradient ≤ 3%, and 3% < gradient ≤ 6% with Channel Migration Zone	Gradient ≤ 1.5%, and 3% < gradient ≤ 6% with No Channel Migration Zone	Gradient > 6%
Tier I Basins	**Prescription "A"**	**Prescription "B"**	**Prescription "C"**
Tier II Basins	**Prescription "D"**	**Prescription "E"**	**Prescription "F"**

[1]Modified from Montgomery and Buffington (1993, 1997), O'Connor and Watson (1998).

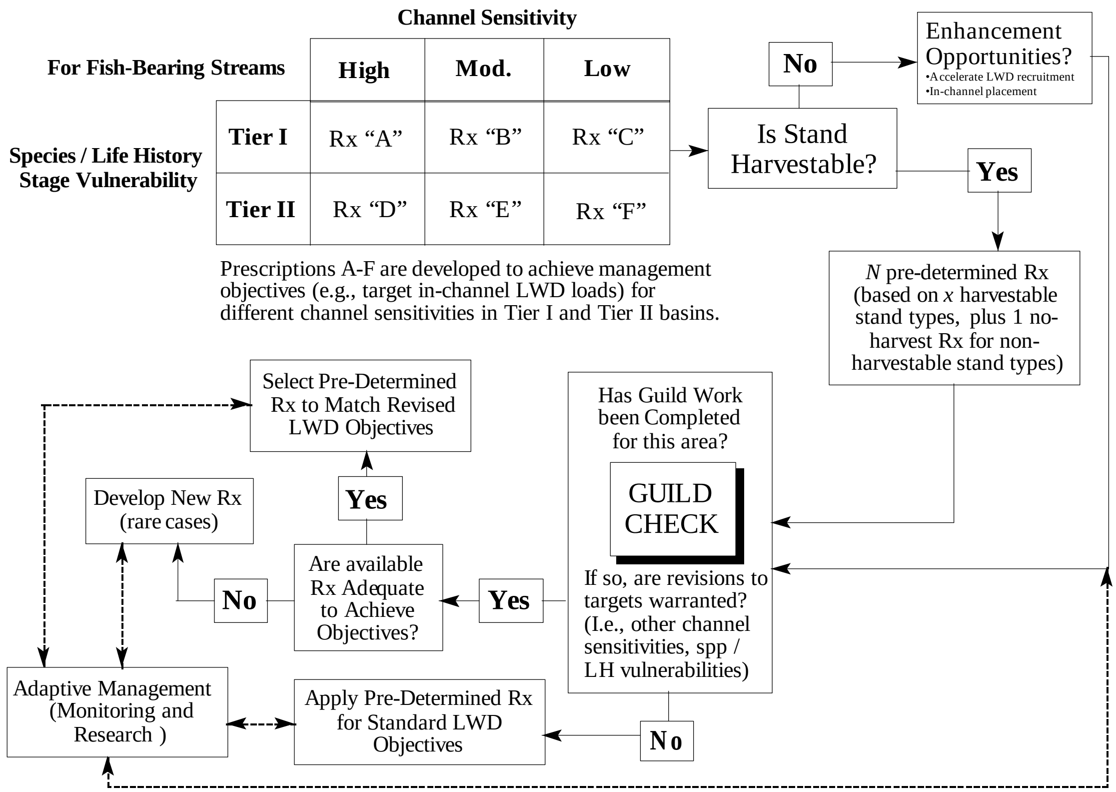
2.4.1 Selecting Riparian Management Options to Address LWD Sensitivities

Once natural patterns of channel and fish habitat response to LWD have been established, an array of management options can be developed to address the different sensitivities. Combinations of different channel types (e.g., plane-bed / forced pool-riffle) and species / life history groups (e.g., Tier I and Tier II) represent discrete "situations" that can be described and mapped. The sensitivity to LWD gain or loss is presumed to differ among these situations. The relative ranking of sensitivities is depicted in Table 5. According to this matrix, loss of LWD in stream reaches of Tier I basins with gradients between 1.5-3% would cause the greatest change in fish habitat, and presumably the greatest potential impact to fish populations. Conversely, relatively high gradient reaches (>6%) occupied by resident cutthroat or other salmonids beside bull trout are expected to be least sensitive to LWD loss. Managers can respond to these different sensitivities with appropriate management options (e.g., prescriptions "A" to "F" in Table 5). This is the "type-specific" notion of riparian protection strategy design (i.e., strategy "C") that was described in the Introduction.

The remainder of this report describes how these concepts will be applied to help select appropriate riparian management options for streams in the project area. Information on the relative LWD-sensitivities of stream reaches, and the conditions of riparian forests must be considered when determining management options. For example, some riparian stands in the project area will not be harvestable, at least in the near term, owing to their existing conditions (e.g., numbers of sub-merchantable trees) and minimum leave tree requirements of state forest practice regulations. For these stands, current conditions may be a result of either natural limitations (e.g., poor growing site), or recent timber harvest. Silvicultural treatments to enhance the LWD recruitment functions of these stands may be warranted (Berg 1995). If LWD recruitment potential was diminished by past practices, then in-channel LWD placement may be appropriate to maintain habitat, or enhance conditions until natural recruitment processes are re-established. The latter will occur when the quantity, size, species, and quality of trees are re-established to sustain LWD inputs to adjacent channels.

For fish-bearing streams, the process of selecting riparian prescriptions is diagramed in Figure 7. The process results in a finite number of prescriptions. Fine-tuning, when necessary, is accomplished through guild assessment (Watson et al. 1998[a]), monitoring, and research (Hicks et al. 1999). In the non fish-bearing portions of the drainage network, a similar process can be used to select riparian prescriptions, but shade/temperature control may be the principal objective in some reaches, instead of LWD. Generally, prescription design can be guided by the riparian protection stratification in Figure 6.

Figure 7. Process for selecting riparian management prescriptions for fish-bearing streams of the project area. *This process assumes one prescription will be developed for each situation, and these will be applied throughout the project area. The stream guilding process is principally used to modify which pre-determined prescription is appropriate for a given situation. Revisions to prescriptions, or creation of new ones, can be accomplished via guilding, research, and monitoring (adaptive management).*



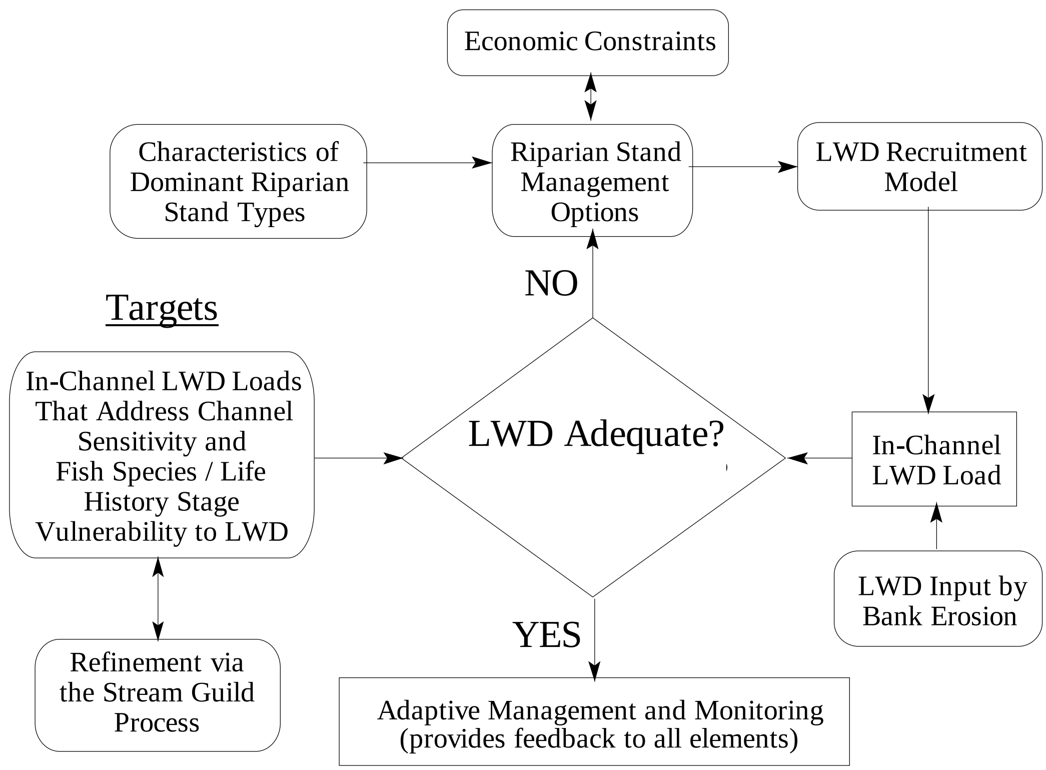
3.0 EVALUATING RIPARIAN MANAGEMENT STRATEGIES

3.1 Development of a Riparian Stand Management Evaluation Process

Riparian Best Management Practices (BMPs) embodied in existing state rules and regulations are designed to protect basic riparian functions (LWD, shade, bank integrity) for the benefit of salmonids. The practices vary, however, and no tools exist to judge their relative, or absolute, effectiveness. Development of such a tool is an important prerequisite to establishing whether additional conservation measures are necessary for protection of native fish species. In this section, a riparian stand management evaluation process for LWD is described, and then applied, to examine outcomes of selected management practices.

Four main ingredients are needed to create a riparian management evaluation tool. First, there must be some form of target condition that relates meaningfully to fish or fish habitat; the target is the yardstick for comparing outcomes of management actions. Second, a means of reliably simulating the response of riparian stands to existing or proposed management actions through time is needed. With this goes the need for realistic information on local riparian stand conditions (tree size and number, for instance). Third, models are needed to translate the stand simulation information into metrics relevant to the targets. Fourth, there must be a process to refine and improve the tool. Together, these ingredients form not only the basis for evaluating, but also for improving, as necessary, the performance of existing protection measures. The flow of information in this stand management evaluation process is illustrated in Figure 8.

Figure 8. Riparian stand management evaluation process developed for use in Plum Creek Timber Company's Native Fish HCP.



3.1.1 Setting Targets

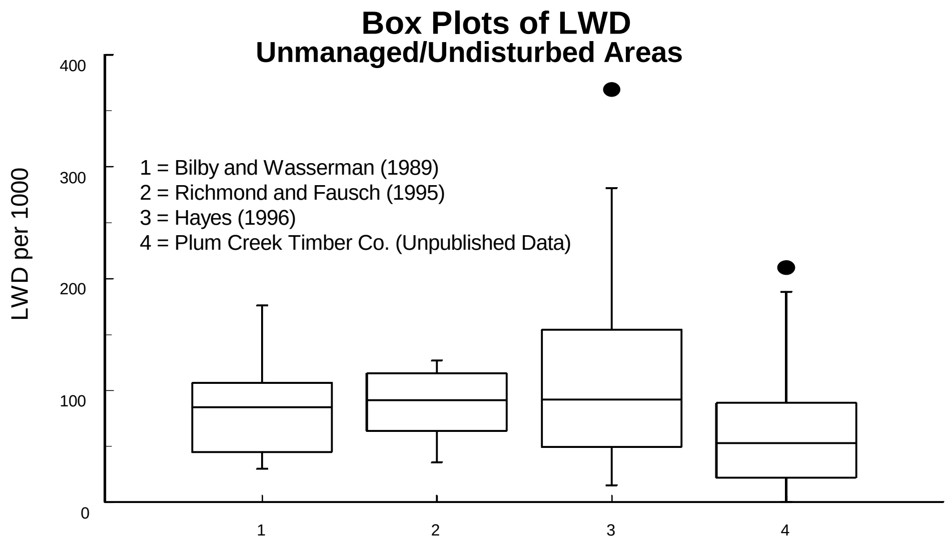
Targets are a crucial component of an evaluation process. Native salmonids evolved with a range of wood loadings in streams, and therefore LWD loads in reference streams should reflect the channel conditions most likely to support and perpetuate healthy fish populations[13]. To judge the performance of various riparian management options, we used in-channel LWD frequencies measured in unmanaged streams of the planning area and other comparable environments. We chose in-channel LWD as our target parameter because it is meaningful to fish, we have relevant reference data for the project area, and because LWD loads can be modeled. Also, the cause-effect relationship between riparian timber harvest, LWD loads, and fish habitat are reasonably well established (see discussion of the Riparian Cause-Effect pathway in the Conceptual Framework section).

[13] Of course, LWD is but one ingredient in the mix of factors that define stream ecosystem structure and function (e.g., Naiman et al. 1992).

We examined the literature and available data to determine which measure of LWD loading would be the most appropriate target. Most research on LWD in streams has been conducted west of the Cascade Mountain Crest (Bilby and Ward 1989, 1991, Beechie and Sibley 1997, Martin et al. 1998, Murphy and Koski 1989). These studies generally indicate that LWD size and density vary according to stream size. In contrast, results of at least one study of LWD in streams east of the Cascade Mt. Crest (Bilby and Wasserman 1989) showed little correlation between LWD size, density, and stream size. This suggested that the two regions would require different targets. Therefore, to examine the relationships between LWD and stream channel characteristics in the less-well studied streams of the Interior Columbia River Basin, we analyzed LWD data from published and unpublished surveys in streams flowing through unmanaged forests. Available data represented a wide geographical area, including the forested mountains of eastern Washington (Bilby and Wasserman 1989), northeastern Oregon (Knight 1990), western Montana (Plum Creek unpublished data, Hayes 1996, USFS (Bitteroot [unpublished data] and Lolo [Riggers et al. 1998] National Forests), Idaho (Huntington 1995, Overton et al. 1996, Philbin 1993), and Colorado (Richmond and Fausch 1995). Data from these studies were compiled and analyzed.

The analyses revealed substantial differences in characteristics of LWD in streams east and west of the Cascade Mt. Crest. Importantly, the relationship between LWD size and frequency east of the Cascade Mountain Crest was found to be independent of stream size. Also, LWD size, frequency, and volume were lower in eastside streams than in westside streams. Of the four studies with comparable methods, LWD loads were roughly similar (Figure 9). These findings greatly simplified target selection, in that a single target of LWD per unit stream length could be used for much of the project area (i.e., eastern Washington, Idaho, and Montana). For convenience, we chose total pieces of LWD per 1,000 ft of stream as the target metric. All pieces with minimum dimensions of 10-cm diameter and 2-m length were selected because these pieces were observed to function as habitat-forming features in streams within the project area (Greg Watson, Plum Creek Timber Company, pers. comm. 1998). Details of this target development work can be found in Appendix B (part1).

Figure 9. Regional comparison of large woody debris (LWD) frequencies in streams flowing through unmanaged riparian forests of the interior Columbia River Basin. Comparable sampling methods were used in these studies.

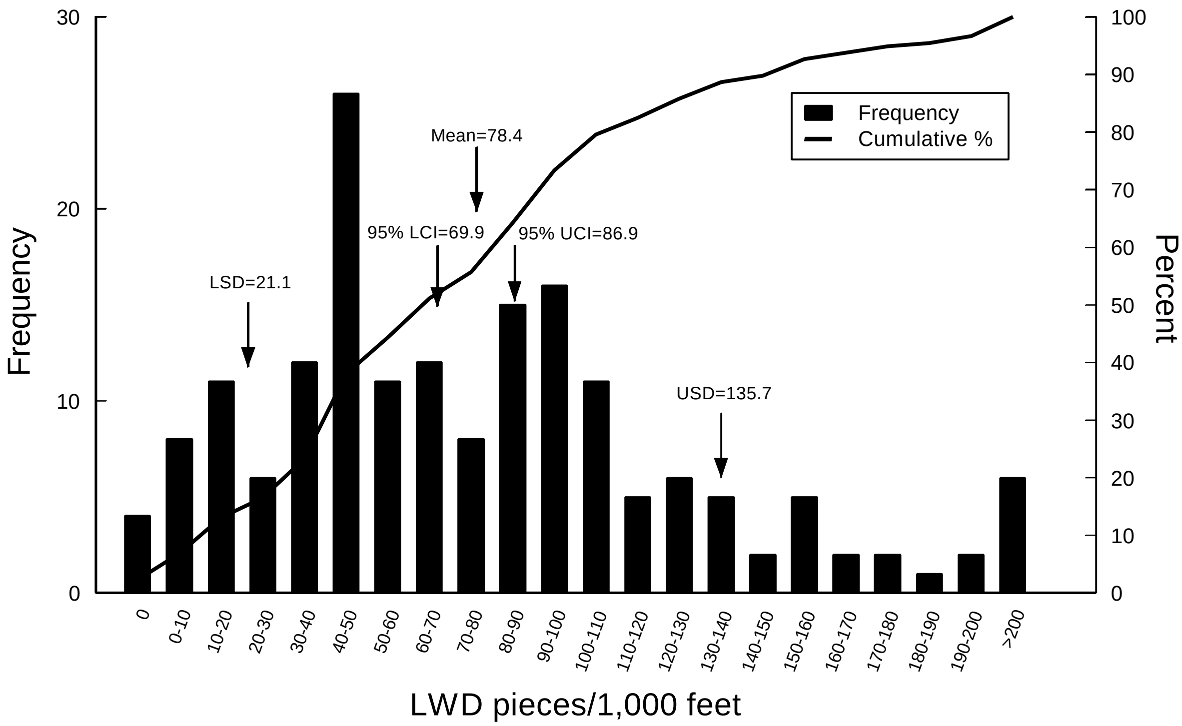


3.1.1.1 Targets for East of the Cascade Mountain Crest

Of the twelve eastside data sets discussed above, four were from studies with comparable methods (e.g., the same minimum dimensions for measured LWD pieces). The studies were: Bilby and Wasserman (1989), Plum Creek Timber Company (unpublished data for 2 separate drainages), Richmond and Fausch (1995), and Hayes (1996) (Figure 9). We pooled these data to derive a target LWD load for use in Montana, Idaho, and eastern Washington. Specifically, a mean LWD frequency of 78 (± 57 S.D.) pieces per 1000 ft of stream length was used as the standard of measurement in the forthcoming evaluations of management options. The distribution of the LWD frequency data behind this target is shown in Figure 10. The target and its supporting data were deemed most appropriate for our purposes because (1) they were pertinent to the project area, (2) we are certain of the methods used (e.g., minimum qualifying piece size), and (3) the minimum piece size is the same as is generated by the LWD recruitment model (RAIS; Welty 1998). The target for fish-bearing waters represents measured values from unmanaged riparian stands along streams between 5 ft and 40 ft wide.

Figure 10. Frequency distribution of LWD loads in streams flowing through unmanaged riparian forests of the interior Columbia River Basin and north central Colorado. *Data are from Bilby and Wasserman (1989), Hayes (1996), Plum Creek Timber Company (unpublished data), and Richmond and Fausch (1995).*

3.1.1.2 Targets for Western Washington



For LWD targets in streams of western Washington, we used data obtained from studies of unmanaged streams in southwest Washington (Bilby and Ward 1989). This was the only study where variability was reported. The average LWD loading from this study is 98 pieces (± 62 S.D.) per 1,000 ft of stream channel (Appendix B, part 1), which we chose as the western Washington target. The average LWD loads reported in other Pacific coastal studies are close to this reference mean (Fausch and Northcote 1992, Harmon et al. 1986). No adjustment was made to the target to account for observed changes in LWD frequency with changing channel width (Bilby and Ward 1991). This is because in the western Washington portion of the project area, most of the streams on Plum Creek Timber Company lands occur higher in the drainage network and are relatively small (~10-ft average bankfull channel width). This channel size is below the range of stream widths studied by Bilby and Ward (1991), so it is uncertain how well the relationship between LWD frequency and channel width applies.

3.1.1.3 Targets for Perennial, Non Fish-Bearing Streams

LWD loads needed to adequately maintain the suite of functions provided by non fish-bearing streams (i.e., sediment storage, streamflow energy dissipation, and nutrient storage/cycling) are even more difficult to determine than are loads needed for fish-bearing reaches where the LWD-habitat relationships are more direct and better understood. However, as with fish-bearing streams, the frequency of LWD in unmanaged non fish-bearing streams is a useful reference. An analysis of the available literature and unpublished data[14] yielded several important conclusions. First, there is a positive relationship between LWD step abundance and stream gradient, with wide scatter in the data, indicating that steeper streams tend to have more LWD steps. Second, although there may be a hint of some regional differences, given the scatter of data, it is reasonable to hypothesize that there are no significant differences in LWD step abundance among regions. This assumption allows available data to be extrapolated to the project area. In the context of channel geomorphology and classification, these data could be interpreted to present potential targets for different channel types based on slope class (Appendix B, part 2, Table B-12). Third, step height is an estimate of sediment storage in LWD steps, which argues for a target that includes a variety of LWD diameters. Fourth, the number of LWD pieces forming functioning steps in a given stream reach was less than the total LWD load in the reach; for unmanaged streams, there were about eight pieces of LWD for every step.

Based on this information, and assuming that most non-fish streams are channels with cascade morphology (>6.5% gradient; Montgomery and Buffington 1997), target LWD loads in non fish-bearing streams would average ~163 pieces per 1,000 ft, and range between 61 and 200 pieces per 1,000 ft. (From Appendix B, part 2, Tables B-12 and B-13). This is a higher wood loading target than for fish-bearing streams, but it accommodates the measured mean number of LWD-formed steps (20 per 1,000 ft), plus an adjustment to account for the total number of LWD pieces needed to produce this many LWD-formed steps (an 8:1 ratio for this gradient range).

3.1.2 Identifying Dominant Riparian Forest Stand Types

The project area contains a great diversity of existing conditions among riparian stands. This is especially true for areas east of Cascade Mountain Crest, where even northern vs. southern aspects of a stream valley can have profoundly different stand characteristics. These differences very likely affect the innate LWD recruitment potential of these stands, as well as the response of these stands to a given management treatment. Therefore, to obtain realistic estimates of LWD inputs as a result of proposed management scenarios, information on actual conditions of stands in the project area were gathered. This also allowed more accurate economic analysis of different management options.

A combination of existing inventories (western Washington) and timber surveys (Interior Columbia River Basin in Montana, Idaho, and eastern Washington) were used to gather information on riparian stands.

3.1.2.1 Riparian Stand Types of the Interior Columbia River Basin

The characteristics of riparian stands in the Interior Columbia River Basin were derived from timber cruises conducted during winter 1998. Data from 640 randomly selected plots in riparian forests of

[14] See Appendix B, part 2 for further details.

the project area east of the Cascade Mt. Crest were used to define dominant stand types. The surveyed stands were diverse and exhibited a wide range of tree sizes, densities, and species. It was, however, feasible and necessary to summarize the inventory data into several forest types for display and for input into a forest growth model. These forest types are defined by quadratic mean diameter (QMD) and the number of trees per acre (TPA) greater than 4.5 feet tall (i.e., diameter breast height; dbh). Table 6 identifies the values of QMD & TPA for each forest type, and Table 7 provides labels for each type. In addition to the forest types listed in the tables there is a non-stocked class representing plots without trees.

This classification effort resulted in nine dominant stand types[15] drawn from a continuum (Figures 11 and 12). The nine stands exhibit distinctive "average" characteristics described in terms of tree density, size, and species. Each has different growth trajectories, and therefore different harvest or enhancement opportunities. Detailed descriptions of each dominant stand type, including visual representations using Stand Visualization Software (Magaughey *in prep*.), are found in Appendix C. Details of survey locations and methods are also provided in Appendix C.

Table 6. Characteristics of dominant riparian stand types on Plum Creek lands in the Interior Columbia River Basin.

	Small (3) 0"<QMD≤6"	**Medium (9) "<QMD ≤12"**	**Large (15) 12"<QMD≤18"**	**Huge (20) QMD >18"**
Low Density (L)	≤ 300 TPA	≤ 200 TPA	≤ 150 TPA	≤ 150 TPA
High Density (H)	300 >TPA ≤ 700	200>TPA ≤ 500	150>TPA ≤ 300	150>TPA ≤ 300
Thick Density (T)	TPA > 700	TPA > 500	TPA > 300	TPA > 300

[15] The "Huge" category in Tables 6 and 7 was not included in the analysis, because the total combined area for these stands represented less than 2% of the streamside miles in the project area.

Figure 11. Characteristics of dominant riparian stand types in Tier 1 watersheds of the Interior Columbia River Basin.

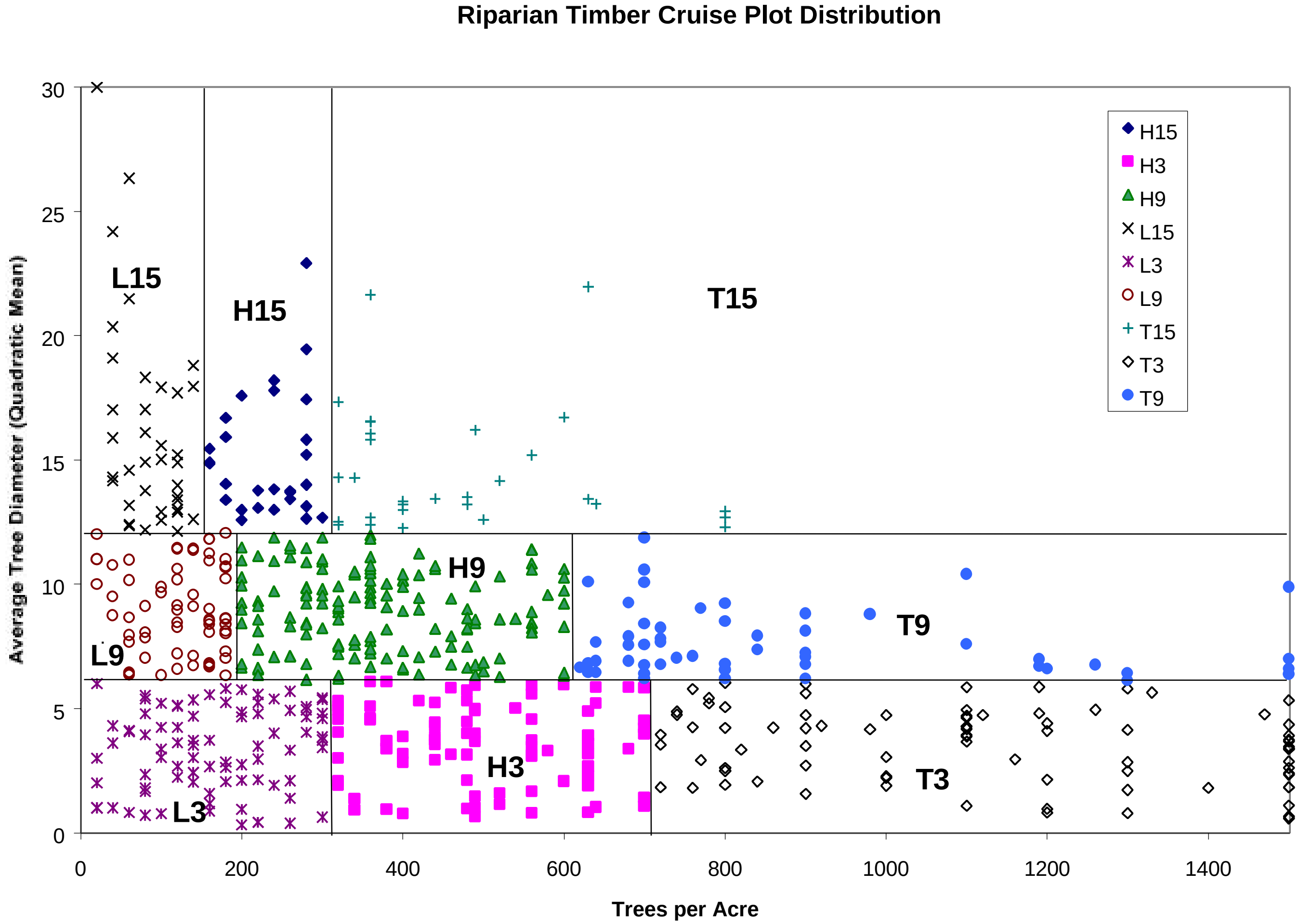


Table 7. Labels for dominant riparian stand types on Plum Creek lands in the Interior Columbia River Basin.

	Small (3)	Medium (9)	Large (15)	Huge (20)
Low Density (L)	L3	L9	L15	L20
High Density (H)	H3	H9	H15	H20
Thick Density (T)	T3	T9	T15	T20

3.1.2.2 Riparian Stand Types of Western Washington

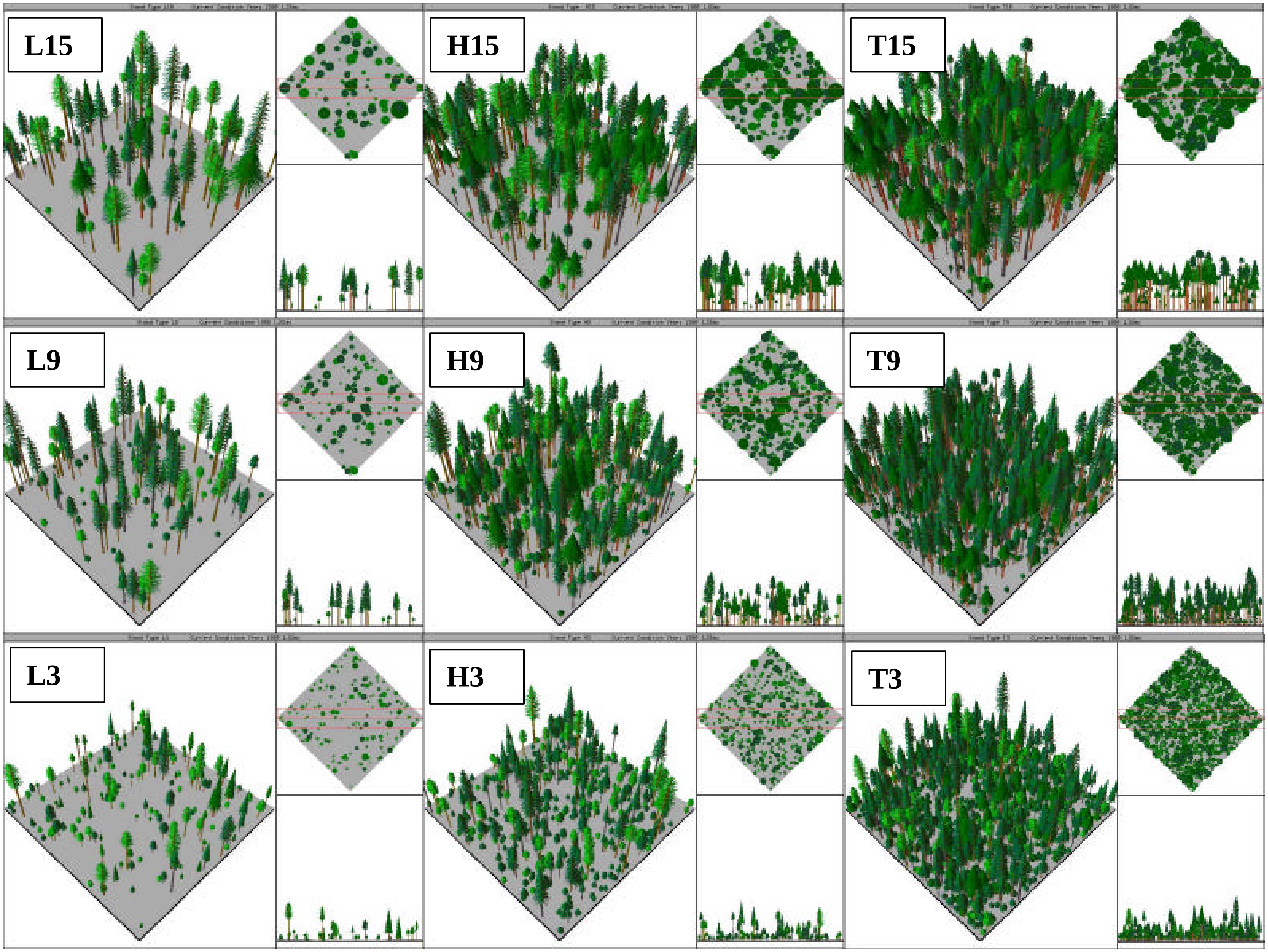
Characterization of riparian stands in western Washington involved air-photo inventory and field reconnaissance. As expected, the composition of westside riparian stands was much less variable than those east of the Cascade Mt. Crest. The dominant tree species were Douglas fir (*Pseudotsuga menziesii*) and red alder (*Alnus rubra*). Virtually all of the riparian areas were composed of second-growth forests. Five dominant stand types were identified based on the age of conifers within ~175 ft[16] of the stream (Table 8). About half of the trees within 50 ft of all streams were alder.

Table 8. Characteristics of dominant riparian stand types on Plum Creek lands in western Washington (Cowlitz River southward). *Red alder (Alnus rubra) were immature in Stand WW1 and mature in all other stands.*

Stand Type	Characteristics	% of Total Stream Miles in Western Washington Planning Area
WW 1	Immature conifer (~ 15 years old) from 0-175 ft	7%
WW 2	Mature conifer (~ 50 years old) from 0-25 ft immature conifer 25-175 ft	17%
WW 3	Mature conifer from 0-50 ft, immature conifer 50-175 ft	26%
WW 4	Mature conifer from 0-100 ft, immature conifer 100-175 ft	12%
WW 5	Mature conifer from 0-175 ft	38%

[16] The estimated Site Potential Tree Height for these Site Class III stands (McArdle et al. 1961).

Figure 12. Visual representation of nine dominant riparian stands in Tier 1 watersheds of the Interior Columbia River Basin.



3.1.3 Simulating the Response of Riparian Forests to Management

3.1.3.1 Forest Vegetation Simulator (FVS)

Forest growth models are used extensively by natural resource managers to estimate future forest conditions. These models, as their name implies, simulate the growth of trees or tree stands using statistically and empirically derived equations. Future tree or stand diameter and height are the main variables predicted by these equations. Mortality is also estimated by most simulators. As in the growth equations, mortality equations estimate diameter and height of dead trees and the number of trees that die. Some simulators estimate the timing, number and size of regeneration. By putting all these functions into a simulator the resource manager has a tool to look at the future and make judgements about what the best management scenario might be to achieve some desired future condition.

The simulation work for Interior Columbia River Basin ("eastside") stands uses the Forest Vegetation Simulator (FVS). FVS is a forest growth simulator developed by the U.S. Forest Service (Wykoff et al. 1982) and used by the Forest Service, forest industry and private landowners. This simulator is described as an individual tree, distance-independent growth model. This simply means individual trees are grown rather than an entire stand of trees. The "distance independent" description tells the user that distances between individual trees are not required to run the simulator. The model was first given the name "Prognosis" and many of the references refer to the model by that name rather than FVS.

The FVS model is one of only a few forest growth models developed, calibrated and tested in the Northern Rocky Mountains. Since the model was developed by the U.S. Forest Service, all of the equations, coefficients and program code are in the public domain and are available for public review and individual user modification. The FVS simulator has several geographical variants for which region specific equations and/or coefficients were developed. We used the unmodified Kookantl variant (version 6.2) in our simulation study. This variant was calibrated with data from the Kootenai and Kaniksu National Forest and the Tally Lake Ranger District of the Flathead National Forest. Information about this variant and the equations used in it can be found on the U.S. Forest Service's Forest Management Service Center's web site at ***www.fs.fed.us/fmsc/variants.html***.

Input data to FVS includes information about individual trees in the form of a tree list. For our simulation, nine tree lists were generated, one for each of the nine riparian forest (stand) types, by averaging all the cruise plot information within each forest type. Each tree list contained the species and number of trees within each 1-inch diameter class. Due to the averaging, several species that appeared in only a few plots showed up in the averaged stand in very small quantities. This sometimes created a stand with species that are not normally found growing together (e.g., Ponderosa pine and Englemann spruce). Removing those species with less than 10% of the total trees kept the composition of these stands in line with commonly observed species mixes and kept the tree list from becoming unnecessarily large and unrealistic. No effect on stand-level growth was anticipated for this data reduction technique. The total number of trees represented by each of the excluded species was proportionally divided into the remaining species.

Stand level input is also required to run the growth simulator. FVS uses habitat type (Pfister et al. 1977) to adjust growth rates across various site qualities. Hansen et al. (1995) identified habitat type associations within riparian areas. We cross-referenced Hansen et al. (1995) and Pfister et al. (1977) to select riparian habitat types that matched model coefficients developed for FVS. Only a few riparian habitat types actually had specific coefficients developed for them, and only two of these were common in the project area. We selected the PICA/EQAR habitat type, number 410 (Pfister et al. 1977) to represent site quality in our simulations.

Each of the nine averaged stands was grown for 100 years in the simulator. Simulator output was reported every decade. FVS output includes an updated tree list and stand summary for each reporting period. The updated tree list contains survivor trees (live trees at both the beginning and end of the simulation), ingrowth trees (new trees that grew during the simulation), and mortality trees (live trees at the beginning and ingrowth trees that died during the simulation). Height, diameter and species at the end of each reporting period are included for each tree. New stand-level values were generated from the output tree list using only 4-inch DBH and larger trees at each reporting period. These new stand values represent the number of trees large enough to qualify as LWD. The new stand values are different from FVS stand level output, which includes trees <4-inch DBH. The new QMD, Lorey height (basal area weighted mean tree height; Husch et al. 1982) and TPA stand values for both live and dead trees were then annualized for each reporting period. This step provided the input data for the large woody debris loading model (see below).

3.1.3.2 The Douglas Fir Simulator (DFSIM)

The simulations for western Washington ("westside") stands were made using the DFSIM model. This is another forest growth simulator developed by the U.S. Forest Service (Curtis et al. 1981) that is commonly used by the Forest Service, forest industry and private landowners west of the Cascade Mountains. Unlike FVS, DFSIM is a whole stand even-aged simulator that projects only Douglas-fir growth and mortality. Stand level attributes, such as stand age, site index, trees per acre and quadratic mean diameter are required for a projection. Future hardwood conditions are derived from alder yield tables (Chambers 1983). Age and site index are used to look up future stand attributes. Mixed hardwood-conifer conditions are predicted by proportionally adding future conifer and hardwood estimates.

3.1.4 Simulating the Effects of Management on In-Channel Large Wood Debris Loads

3.1.4.1 Riparian Aquatic Interaction Simulator

Outputs from the FVS and DFSIM growth models were used as inputs to the "Riparian Aquatic Interaction Simulator" (RAIS; version 3.0, Welty 1998). This model estimates in-channel LWD loads through time by incorporating such factors as depletion rate, channel width, riparian buffer width, initial LWD loading, etc. These factors are entered interactively at the start of each simulator run.

LWD estimates and methodology in RAIS are based on simple principles of geometry and on published riparian-LWD research (Bilby and Ward 1989; 1991, McDade et al. 1990, Murphy and Koski 1989, Rainville et al. 1985, Robison and Beschta 1990, Van Sickle and Gregory 1990, Welty 1998). Annual mean effective stand height[17] is computed using mortality QMD and Lorey height. The buffer width (defined by the user) is divided into ten equal strips and the distance from the midpoint to the channel edge is determined for each strip. With mean effective stand height and distance to the channel known, the random probability of an average-sized mortality tree falling into the channel[18]

[17] Effective Height is the height of the tree where the stem diameter is equal to the minimum qualifying LWD diameter; i.e., 4 in. This height is computed from a tree taper function (Rustagi and Loveless 1991).

[18] To qualify as LWD, a 4 in. diameter piece must protrude beyond the bank a minimum of ~6.5 ft).

can be calculated using the equation from Van Sickle and Gregory (1990). Each of the ten probabilities, one for each strip, is multiplied by one tenth (1/10) of the mortality, to estimate the LWD input (minimum qualifying piece size = 10 cm x 2 m) for each strip. Summing the ten estimated LWD inputs produces the total LWD input for a given forest type. This procedure is applied to each year's mortality, using the appropriate mortality QMD and Lorey height values, to generate estimated annual LWD inputs.

Annual LWD inputs are accumulated by the simulator. A depletion rate is multiplied by the annual accumulated LWD value to account for LWD that rots or is washed away during high flows (Murphy and Koski 1989). This product is then subtracted from the accumulated value to produce estimated in-channel LWD piece counts at each year. The in-channel LWD piece counts are then periodically reported back to the user.

For each timber harvest option described in the next section, the RAIS simulator was run separately for different channel widths. The RAIS model output is sensitive to stream width because the depletion rate (from Murphy and Koski 1989) increases with channel size. For eastside streams, channels ten and thirty feet wide were used. These widths bracket the sizes of most fish-bearing streams in the eastside portion of the project area. For western Washington, an average 10-ft wide channel was used. Other important user-defined settings for RAIS simulations were:

1. Initial in-channel LWD loads of one-half the mean values for unmanaged streams in eastside and westside forests (see Appendix B, Part 1). This loading level was chosen to reflect a hypothesized lower abundance of LWD in streams where riparian forests were harvested historically without regard for LWD recruitment. Although several studies have shown no difference in LWD loads in managed vs. unmanaged streams of the Interior Col. R. Basin (e.g., Bilby and Wasserman 1989, Huntington 1995, Plum Creek Timber Company, unpublished data), such differences have been observed in western Washington (Bilby and Ward 1989). Lack of detectable LWD losses in streams where riparian forests were once clearcut may simply reflect a time lag between interruption of inputs and depletion of residual LWD;
2. Depletion rate of 1.98% per year for minimum qualifying LWD pieces (10 cm x 2 m) (Murphy and Koski 1989) and;
3. Random direction of tree fall (Van Sickle and Gregory 1990).

3.1.4.2 Estimated LWD Inputs from Bank Erosion

The RAIS model estimates LWD inputs largely from trees that fall after succumbing to competition (stand-suppression) mortality[19]. We suspect that other LWD recruitment processes, such as bank erosion, disease, fire, and windthrow, have equal or greater importance as LWD delivery mechanisms (Agee 1994, Benda and Sias 1998, Keller and Swanson 1979, Murphy and Koski 1989).*Reliance on FVS-modeled mortality as the sole LWD recruitment mechanism was therefore likely to underestimate LWD inputs*. To partly compensate, we included estimated LWD inputs from bank erosion. The estimates for bank erosion are developed below based on available literature values, our knowledge of typical riparian and stream conditions for the project area, and specified assumptions.

Despite considerable uncertainty regarding actual bank erosion rates, a reasonable minimum bank erosion rate from Hooke (1980) can be used as a first approximation to assess possible LWD recruitment rates by this source. After field measurement and literature review, Hooke (1980) concluded a minimum average bank erosion rate was about 5 cm/yr (0.16 ft/yr). He also found that bank erosion rates could be predicted as a function of drainage area.

A bank erosion rate of 0.16 ft/yr is equivalent to an average of 4.8 ft of bank retreat over the proposed thirty year life of the HCP. Based on field observations in streams of the Swan and Thompson watersheds in Montana, this rate appears reasonable for relatively low-gradient channels with at least modest stream power (i.e. drainage area for at least a 2nd or 3rd order channel). However, bank erosion processes may not operate uniformly, and considerable variation is to be expected. These channels typically have a well-developed floodplain and/or terrace, suggesting that alluvial processes are active, and that persistent bank erosion is likely. This hypothesis fits the definition of alluvial streams in which floodplain and/or terrace deposits are accumulated by over-bank deposition and depleted by bank erosion in response to long-term variation in the ratio of sediment transport to sediment supply (Richards 1982). The bank erosion rate of 0.16 ft/yr is likely to be too high for smaller, more confined channels and channels with bedrock or boulder banks.

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[19] RAIS allows for user-specified inputs from windthrow, but this option wasn't used because the extent of windthrow as a LWD recruitment process is not well understood in the project area. RAIS also allows users to incorporate deliberate placement of LWD, but this option wasn't used because the goal of this effort is to evaluate the effect of various management options on natural recruitment processes.

3.1.4.2.1 Hypothetical LWD Recruitment by Bank Erosion

3.1.4.2.1.1 Bank Erosion Estimates for streams of the Interior Columbia River Basin

The estimated LWD recruitment from bank erosion for each of the nine Interior Columbia River Basin ("eastside") riparian stand types using the 0.16 ft/yr bank erosion rate is summarized in Table 9. For these estimates, riparian forest stand density was first assumed to be 100 TPA, distributed in a 25-ft wide strip along 1000 ft of channel. On one bank with an area of 25,000 ft^2 (0.574 ac), this represents 57 trees per 1000 ft or 5.7 trees per 100 ft. Assuming the trees to be uniformly distributed, and allocating a proportional number of stems in 5-ft wide strips parallel to the stream bank, then there would be 11.4 stems in a 5-ft wide strip adjacent to 1,000 ft of channel on one bank. Within the

Table 9. Estimated inputs of LWD from bank erosion based on an average bank erosion rate of 0.16 ft./yr, for one side of the stream. *Estimates are given for eastside and westside riparian stand types and LWD diameter classes (eastside only), based on data collected within 50 ft of the stream channel.*

Stand Type	Recruited Trees / 1000 ft / decade				Number of Recruited Trees Remaining After 30 years (assuming 1.5% per year depletion rate; nearest whole piece)
	Tree Diameter Class				
	4-12 in	13-20 in	>20 in	Total	
L3	2.1	0.1	0	2.1	5
L9	2.3	0.8	0.2	3.2	7
L15	1.2	1.1	0.6	2.9	6
H3	5.0	0.3	0	5.4	12
H9	6.7	2.3	0.5	9.5	20
H15	3.5	2.2	1.7	7.4	15
T3	12.5	0.8	0	13.3	29
T9	14.6	3.4	0.9	18.9	37
T15	7.4	4.3	2.7	14.3	25
Western Washington Stands					
WW1					15
WW2 – WW5					11

first 5-ft strip adjacent to the channel, 11.4 trees uniformly distributed in positions perpendicular to the bank are spaced at 0.44 ft. Consequently, one tree would be recruited every three years along a 1000-ft reach. Over a 30-year period, bank erosion would total about 5 ft and about 11 stems would be recruited. Based on the assumptions above, for a 100 TPA stand, the LWD recruitment rate is estimated to be 0.00038 stems/ft of stream bank/yr.

3.1.4.2.1.2 Bank Erosion Estimates for Western Washington Streams

The same relative erosion rates were used to estimate LWD inputs from bank erosion in western Washington ("westside") streams. For immature stands (WW1), no trees were assumed to be large enough to enter the channel before ten years. Afterward, estimated tree size and density data from DFSIM growth simulations were used to calculate the number of trees recruited to the channel via bank erosion. With depletion, approximately 15 pieces of qualifying LWD (10 cm x 2 m) per 1000 ft were estimated to remain in the channel after thirty years. For mature forests (Stand Types WW2-WW5) recruitable trees are available immediately and from these an estimated 11 trees/1000 ft remain in the channel after thirty years. Lesser inputs from mature stands reflect the lower tree densities and lower mortality in these older forests. Growth simulations (DFSIM) do not provide diameter classes, therefore for western Washington streams only the number of LWD pieces are presented in Table 9.

3.1.4.2.2 Observed LWD Recruitment via Bank Erosion and other Processes

Data for LWD recruitment processes are not widely available. This section summarizes results from Murphy and Koski (1989) for southeast Alaska and from observations of LWD in the Swan River Valley watershed analysis. Data from the Swan R. Valley were collected by the consulting firm of O'Connor Environmental, Inc. (OEI), in connection with the stream channel assessments of the Goat/Sqeezer creek, and Piper Creek Watershed Analyses (Watson et al. 1998[b]).

Murphy and Koski (1989) report LWD input processes for inventoried LWD. Their data are percentages of inputs from known processes; this ignores the majority of LWD pieces for which an input mechanism cannot be inferred in the field. Murphy and Koski's (1989) data for channels described as alluvial are summarized in Table 10, along with OEI data for alluvial channels in the Swan Valley.

Table 10. Percentage of LWD inputs attributed to input processes for those LWD pieces where input process can be determined based on field observations. *These data represent a minority of LWD pieces inventoried; input sources for most LWD cannot be determined. Whether these data are representative of all LWD inputs is not known.*

Input Process	Southeast Alaska*	Swan Valley, Montana**
Bank Erosion	55	68
Windthrow	28	20
Mortality	16	n.a.
Landslides	0	0
Logging Debris	n.a.	12

*Murphy and Koski (1989), **O'Connor Environmental Inc., unpublished data.

From Table 10, it appears that bank erosion is the dominant input process in southeast Alaskan alluvial streams. The data from Montana cannot be interpreted in this way, as LWD inputs by "mortality" were not determined. Both inventories suggest that LWD inputs by bank erosion are two to three times greater than by windthrow. It should be noted that LWD recruitment by bank erosion is not expected to be uniform throughout the drainage network. Bank erosion potential is greatest for larger, lower gradient streams (e.g., <3%). LWD inputs from bank erosion are also expected to be relatively significant in higher gradient (3-6%) reaches with channel migration zones.

Together, LWD recruitment from stand-suppression mortality and bank erosion represent quantifiable inputs that each stand is estimated to deliver to an associated channel over time. In the next section, we use this information to examine the effects of various timber harvest options on LWD loads.

3.2 Application of the Riparian Stand Management Evaluation Process

3.2.1 Fish-Bearing Streams East of the Cascade Mountains (Montana, Idaho, and Eastern Washington)

The effect of different management options on LWD loads after thirty years were simulated using a combination of the models described above. For eastside fish-bearing streams, eight different management options were evaluated, each with increasing numbers of post-harvest residual trees. The scenarios were based on existing Montana law which allows harvest of up to 50% of the trees >8" dbh, to a minimum stocking of 87 trees per acre (TPA) >8". The phrase “allowable harvest” used below therefore refers to what Montana state regulations would currently allow (i.e., 1999). The options ranged from complete tree removal (clearcut) to no harvest within a distance equal to a site potential tree height (100-ft). All harvested trees were taken from within 50 ft of the channel. Specific options were:

1. Clearcut to the edge of the stream;
2. 100% allowable harvest uniformly distributed within 50 ft of the channel;
3. 100% allowable harvest, with 25% harvested in the first 25 ft, and 75% harvested in the second 25 ft;
4. No harvest in the first 25 ft, and 100% allowable harvest in the second 25 ft;
5. No harvest in the first 25 ft, and 50% allowable harvest in the second 25 ft;
6. No harvest within 50 ft of the stream channel;
7. No harvest within 75 ft of the stream channel; and
8. No harvest within 100 ft of the stream channel.

The clearcut option represents historical practices. The uniform harvest option (#2) was considered to represent legal limits according to existing Montana SMZ rules, whereas the staggered harvest option (#3) was considered to best reflect SMZ rules as they are typically practiced by Plum Creek foresters, on Class I streams. For all simulations, at least 87 TPA were left within 50 ft of the channel post-harvest, to comply with Montana’s existing SMZ requirements. Simulations assume that riparian timber harvest occurs in all stands immediately and only once during the 30-yr timeframe of the HCP.

Table 11. Estimated in-channel LWD loads* after 30 years for each eastside stand type under different management options. *For all options, channels were assumed to start with 39 pieces of LWD (minimum dimensions of 10 cm diameter x 2 m length) per 1000 ft of stream. This is half the average LWD load found in unmanaged streams of the Interior Columbia River Basin (see text). Values shown are pieces of LWD per 1000 ft of stream channel. Ratios shown in parentheses represent the percentage of merchantable trees removed in the first and second 25-ft zones adjacent to the stream, based on Montana's existing streamside management guidelines. No LWD inputs from bank erosion are included.*

Stand Type	Option 1 Clearcut to edge of stream	Option 2 (50:50)	Option 3 (25:75)	Option 4 (0:100)	Option 5 (0:50)	Option 6 (No cut 50 ft)	Option 7 (No cut 75 ft)	Option 8 (No cut 100 ft)
LWD Loads in a 10-ft Wide Channel after 30 years								
L3	21	N/A**	N/A	N/A	N/A	22	22	22
L9	21	N/A	N/A	N/A	N/A	23	23	23
L15	21	N/A	N/A	N/A	N/A	25	25	25
H3	21	N/A	N/A	N/A	N/A	25	25	25
H9	21	33	48	52	55	65	67	67
H15	21	48	49	60	68	84	89	89
T3	21	N/A	N/A	N/A	N/A	26	26	26
T9	21	98	102	140	154	182	187	187
T15	21	92	107	142	167	222	243	243
LWD Loads in a 30-ft Wide Channel after 30 years								
L3	19	N/A	N/A	N/A	N/A	19	19	19
L9	19	N/A	N/A	N/A	N/A	20	20	20
L15	19	N/A	N/A	N/A	N/A	22	22	22
H3	19	N/A	N/A	N/A	N/A	19	19	19
H9	19	30	36	40	41	49	49	49
H15	19	41	45	55	60	67	71	71
T3	19	N/A	N/A	N/A	N/A	27	27	27
T9	19	64	65	82	92	114	114	114
T15	19	73	93	127	140	166	181	181

*Estimates only include LWD inputs from stand-suppression mortality.

**N/A = Not Applicable. These stands do not contain enough trees > 8" dbh to allow harvest.

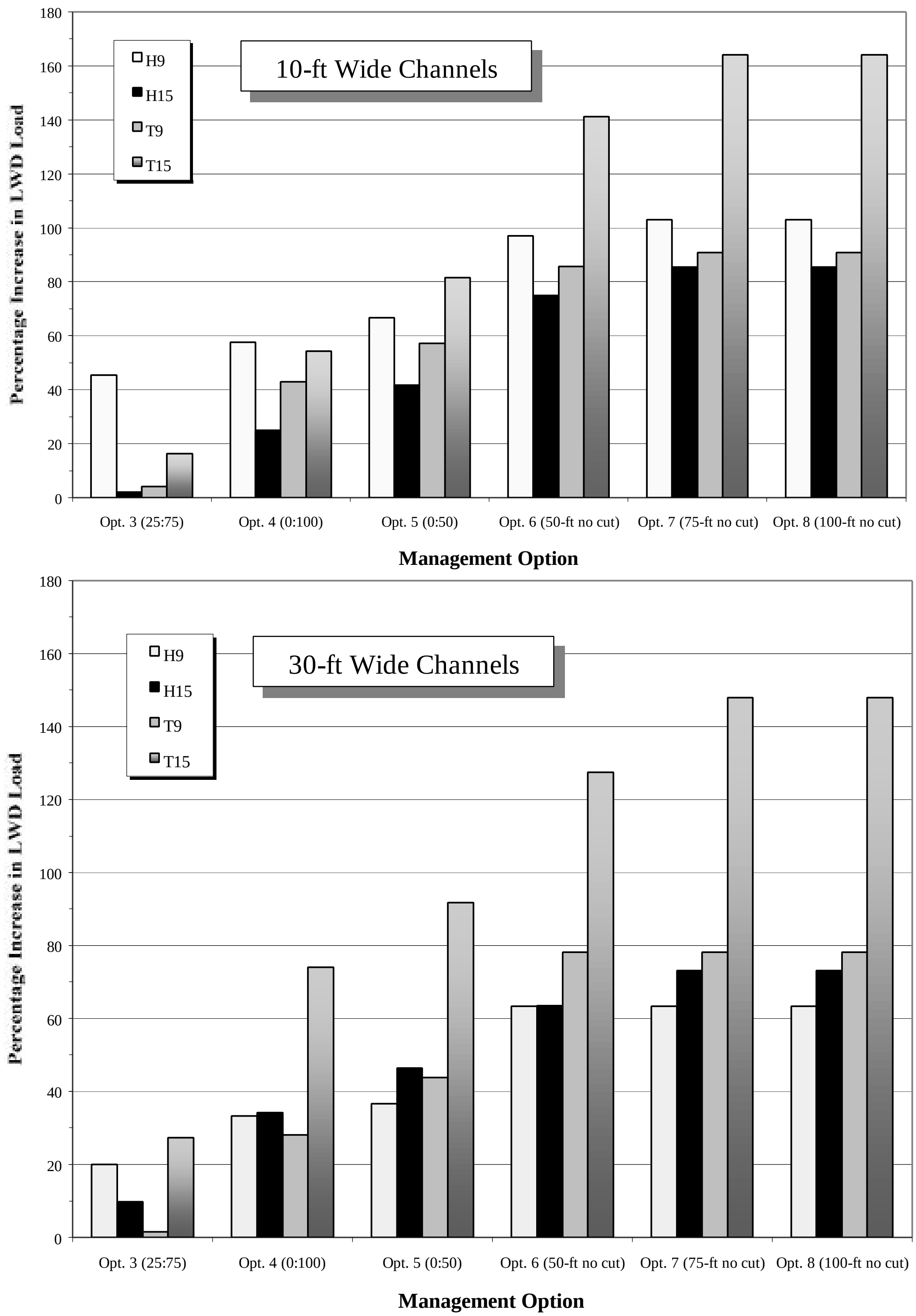
3.2.1.1 A Snapshot in Time: LWD Loads 30 Years After Harvest

Six general conclusions can be drawn from the results of the simulated management options. First, only four of the nine riparian stand types contain enough trees >8" dbh to allow immediate or near-term harvest (Table 11). Second, the complete interruption of LWD inputs by clearcutting to the stream's edge (option 1) creates a situation where LWD loads are not restored after 30 years regardless of stand type. Third, the low density stands (L3, L9, L15), and the high density stands with small trees (H3, T3) contribute little LWD within 30 years even when left unmanaged (options 6-8). Fourth, only marginal amounts of LWD (0-10%, depending on stand type and stream size) come from beyond 50 ft. Fifth, for the harvestable stands (H9, H15, T9, T15), as the number of leave trees within 50 ft increases, the LWD loads increase. Sixth, differences in outcomes among management options can be seen after thirty years (as opposed to western Washington stands; see westside section below).

Timber harvest within 50 ft of the stream has the greatest influence on LWD recruitment. Relative to current legal limits (i.e., Option 2, where allowable harvest is evenly spread throughout the 50-ft SMZ), LWD loads can be increased by 25% to 75% simply by adjusting where trees are cut within the SMZ (compare options 3 and 4 in Figure 13). Reducing the allowable harvest by 25% overall, and concentrating the harvest in the outer 25-ft of the SMZ (i.e., option 5) can result in as much as a 92% increase over current practices, depending on stand type and stream size (minimum increase = 37% for H9 stands). Precluding timber harvest within 50-ft of stream channels (option 6) can boost LWD loads after 30 years by 63% to 141%. Applying a no-harvest prescription to the area between 50 and 75 ft would result in zero to 23% more LWD than option 6 after 30 years. No additional LWD is added by widening the no-harvest zone beyond 75 ft.

Managers can achieve a range of outcomes with potentially small additional cost by adjusting practices within 50-ft of eastside streams. Conversely, trees left beyond 50-ft will contribute proportionately little LWD to streams. Because stand types influenced the outcomes, design of management practices could be refined by incorporating stand type information. This would improve "smart buffer" design to address the different channel sensitivity strata described in section 3.4. Stream size is another factor that could be considered in buffer design, however, the different outcomes for 10-ft and 30-ft channels were caused by different LWD depletion rates, and these have not been substantiated in eastside streams.

Figure 13. Increases in woody debris loads for different management options in eastside riparian stands after 30 years, relative to outcomes under Montana's existing SMZ guidelines.



3.2.1.2 Long-Term Trends in LWD Loads

Timber harvest and other activities within riparian areas can have long-lasting consequences with respect to functions that influence aquatic ecosystems. Therefore it is appropriate to consider the outcomes of management actions both within and beyond the 30-year HCP timeframe. Further insight into the relative performance of each management option with respect to target LWD loads can be gained by examining trends in LWD loads through time. For this, a 100-year outlook[20] is provided for each of the harvestable stands (Figures 14 and 15 for 10 and 30 ft channels, respectively). For the first 30 years (i.e., the timeframe of the HCP), LWD loads increase or are stable. One exception is option 2 (50:50) which shows an approximate 10% decrease over the plan term for stand type H9 only.

When compared to target LWD loads, the levels of timber harvest in all management options[21] (i.e., options 2-5) produce and maintain LWD loads that are within one standard deviation of the target mean[22] within and beyond thirty years. In general, H9 and H15 stands produce LWD loads that are at or below the mean, whereas T9 and T15 stands produce loads at or above the mean. Trends are similar for small and large channels, but LWD loads in larger channels are lower for all stand types and options. Long-term LWD trends also show no dramatic increases or decreases relative to general trends in LWD loads through time for all stand types. Notwithstanding the potential limitations of the forest growth models used to create these riparian trajectories, the trends suggest that the response of riparian forests seen in the 30-years of the HCP will be echoed for many decades afterward.

[20] The accuracy of the *absolute* LWD load projections this far into the future is questionable. However, the *relative* LWD loads should be reasonable.

[21] Clearcutting to the stream's edge is not considered a viable harvest option, and was merely used to illustrate the potential effect of past practices. This option will not be considered further.

[22] From surveys of eastside streams with unmanaged riparian forests; 78±57 LWD pieces per 1000 ft.

Figure 14. Estimated trends in total woody debris loads in Interior Columbia River Basin streams by stand type. *LWD loads represent pieces per 1,000 ft. of stream channel with minimum qualifying dimensions of 10 cm.diameter x 2 m. length. Average channel width =* ***10 ft.*** *Reference mean ± SD reflects pooled data from comparable surveys of LWD loads in unmanaged streams of the Interior Columbia River Basin (Bilby and Wasserman 1989, Hayes 1996, Plum Creek Timber Company unpublished data) and north-central Colorado (Richmond and Fausch 1995). See text for descriptions of stand types and management options.*

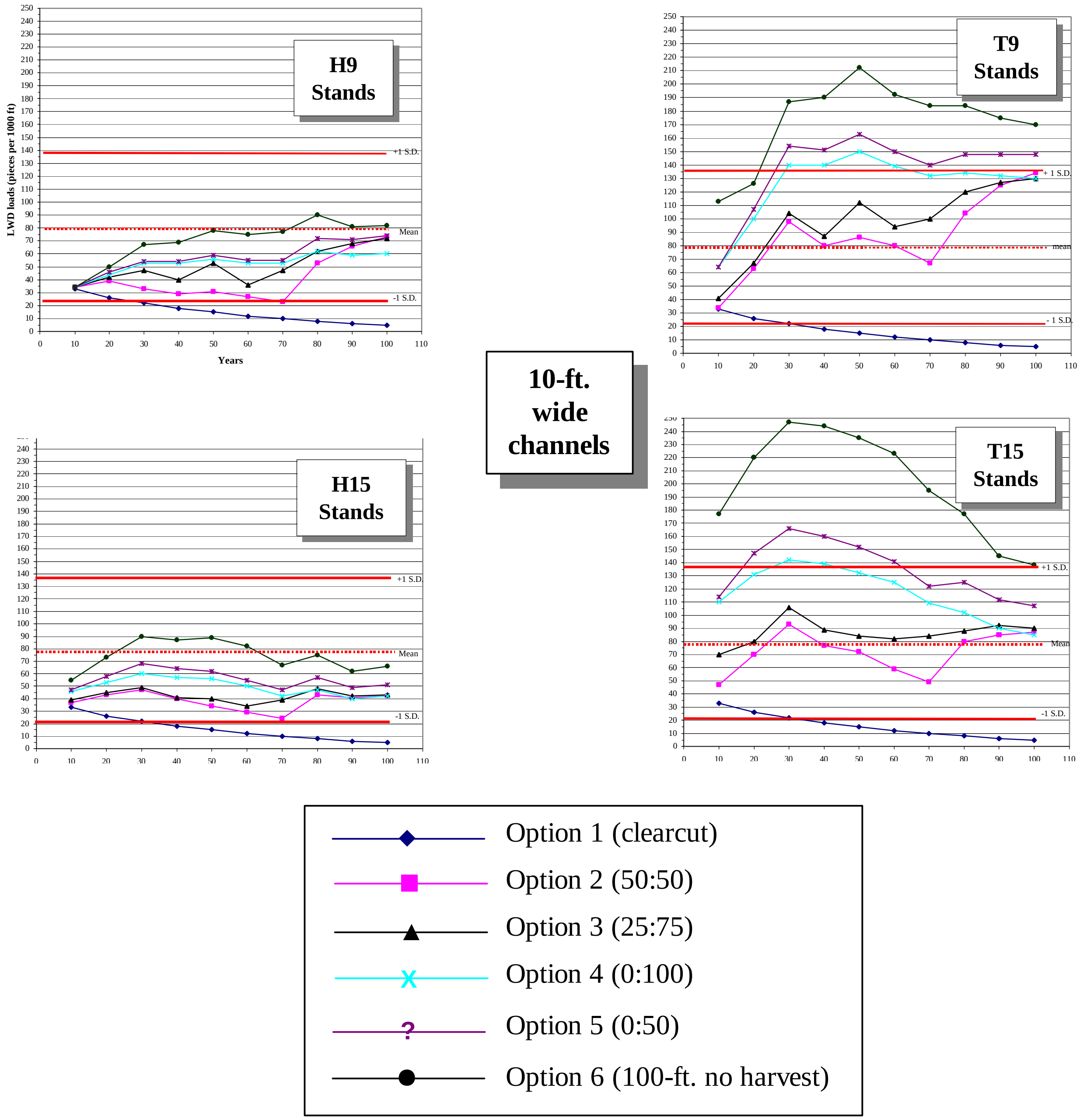
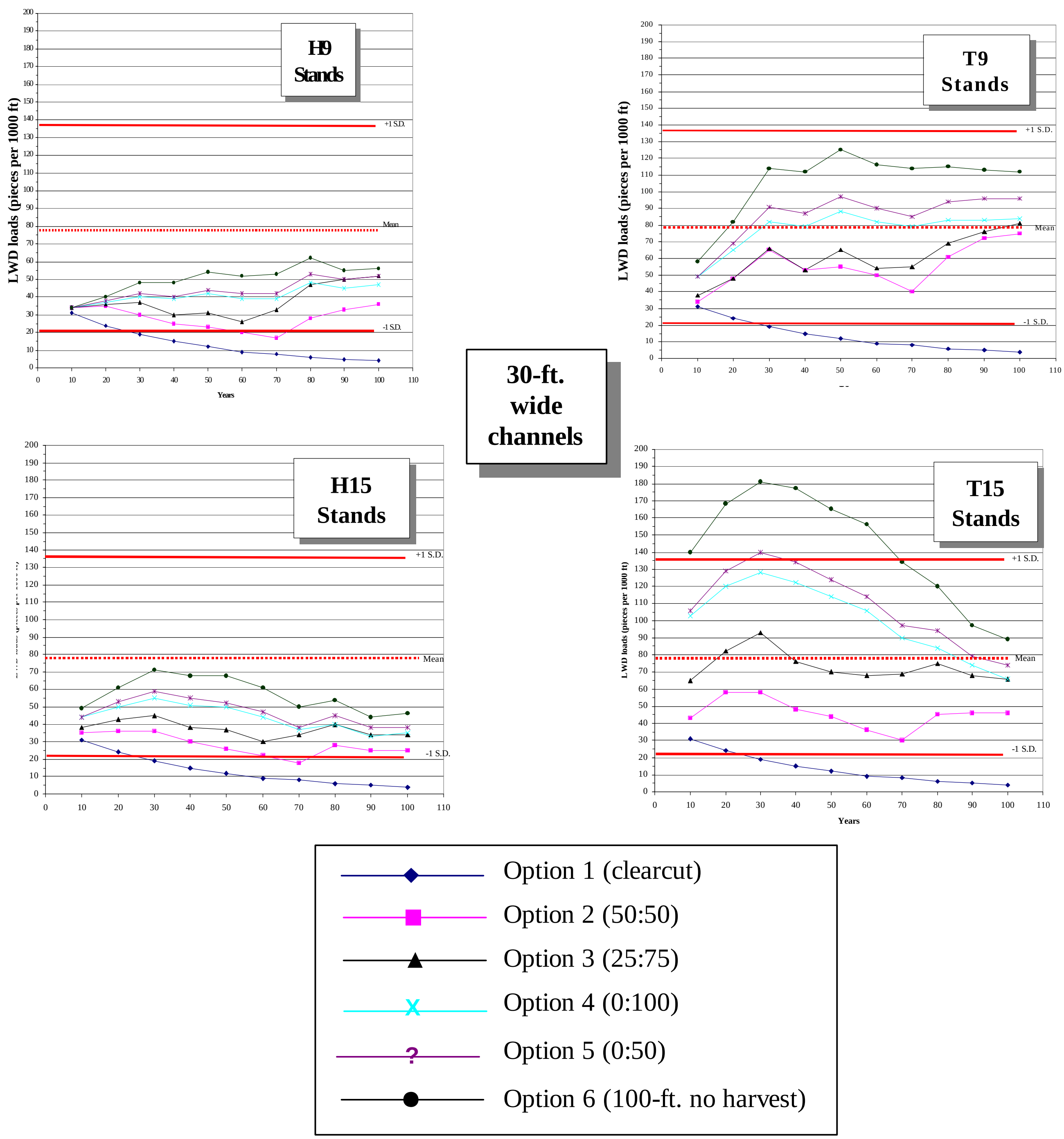


Figure 15. Estimated trends in total woody debris loads in Interior Columbia River Basin streams by stand type. *LWD loads represent pieces per 1,000 ft. of stream channel with minimum qualifying dimensions of 10 cm. diameter x 2 m. length. Average channel width =* ***30 ft.*** *Reference mean ± SD reflects pooled data from comparable surveys of LWD loads in unmanaged streams of the Interior Columbia River Basin (Bilby and Wasserman 1989, Hayes 1996, Plum Creek Timber Company unpublished data), and north-central Colorado (Richmond and Fausch 1995). See text for descriptions of stand types and management options.*



3.2.1.3 Effect of Bank Erosion on LWD Loads in Streams of the Interior Columbia River Basin

Thus far, LWD inputs from bank erosion have not been included in the analyses. These inputs add significantly to the LWD loads derived solely from stand-suppression mortality. For illustration, LWD loads after 30 years are shown in Table 12 with and without bank erosion inputs, with differences expressed as a percentage increase in LWD loads.

For unmanaged stands, bank erosion contributes 10-111% more LWD than stand suppression alone for 10-ft channels, and 14-107% more for 30-ft channels. The increases are most pronounced for stand types H3 and T3 that contribute little stand-suppression derived LWD owing to small trees. Among the harvestable stands, the effect of bank erosion on managed stands is approximately two- to three-times the increases seen for the no-harvest option. Bank erosion inputs are important when evaluating management options against absolute LWD targets. The addition of the estimated inputs improves the performance of all options.

Table12. Effect of bank erosion on LWD loads after 30 years for eastside stand types. *The relative contribution of LWD inputs from bank erosion is illustrated by comparing LWD loads from managed and unmanaged riparian areas. Bank erosion is most likely to contribute significant amounts of LWD in larger, lower gradient (<3%) stream channels, or in steeper channels (3-6%) with channel migration zones. See text for further discussion.*

Stand Type	LWD Loads[1] in a 10-ft wide channel after 30 years					
	50-ft Riparian Management Zone with (Montana SMZ guidelines; i.e., Option 2 from Table 10), and commercial (partial) harvest beyond 50 ft			100 ft No Harvest Riparian Management Zone		
	Without Bank Erosion	With Bank Erosion	Percent Increase in LWD loads	Without Bank Erosion	With Bank Erosion	Percent Increase in LWD loads
L3	N/A[2]	N/A	N/A	22	27	23%
L9	N/A	N/A	N/A	23	30	30%
L15	N/A	N/A	N/A	25	31	24%
H3	N/A	N/A	N/A	25	37	48%
H9	33	53	61%	67	87	30%
H15	48	63	31%	89	104	17%
T3	N/A	N/A	N/A	26	55	111%
T9	98	135	38%	187	224	20%
T15	92	117	27%	243	268	10%
LWD Loads[1] in a 30-ft wide channel after 30 years						
L3	N/A	N/A	N/A	19	24	26%
L9	N/A	N/A	N/A	20	27	35%
L15	N/A	N/A	N/A	22	28	27%
H3	N/A	N/A	N/A	19	31	63%
H9	30	50	67%	49	69	41%
H15	41	56	36%	71	86	21%
T3	N/A	N/A	N/A	27	56	107%
T9	64	101	58%	114	151	32%
T15	73	98	34%	181	206	14%

[1]Piecesof LWD (10 cm diam. x 2 m length) per 1,000 ft of stream channel.

[2]N/A = Not Applicable. These stands do not contain enough trees > 8" dbh to allow harvest.

3.2.2 Non Fish-Bearing Streams East of the Cascade Mountain Crest (Montana, Idaho, and Eastern Washington)

Simulated management options for non fish-bearing streams were somewhat different than for fish-bearing streams (see "setting targets" section above). The aim of the evaluation was to discover whether a continual supply of LWD was being delivered to streams through the plan term, and how well the simulated LWD loads compared with targets. Therefore, the evaluation centered on the outcomes of management according to Montana's existing SMZ guidelines. These guidelines are presently followed on the majority of non fish-bearing streams in the planning area. They call for leave trees within the same 50 ft core LWD-generation area as do rules governing harvest along fish-bearing streams. The primary difference is that some of the non fish-bearing streams in Montana allow a 50% harvest down to a lower minimum tree retention level. Non fish-bearing perennial (Class 1) streams receive an 87 TPA minimum retention. Intermittent streams and perennial streams that are not connected by surface flow to a fish-bearing stream are designated as Class 2 streams. The minimum tree retention for Class 2 streams is 44 TPA >8"dbh, representative of size and species of the pre-harvest stand. The same general conclusions noted earlier for fish-bearing streams are expected to apply for this class of stream type (e.g., greater LWD loads are produced with increasing numbers of leave trees within 50 ft of the channel).

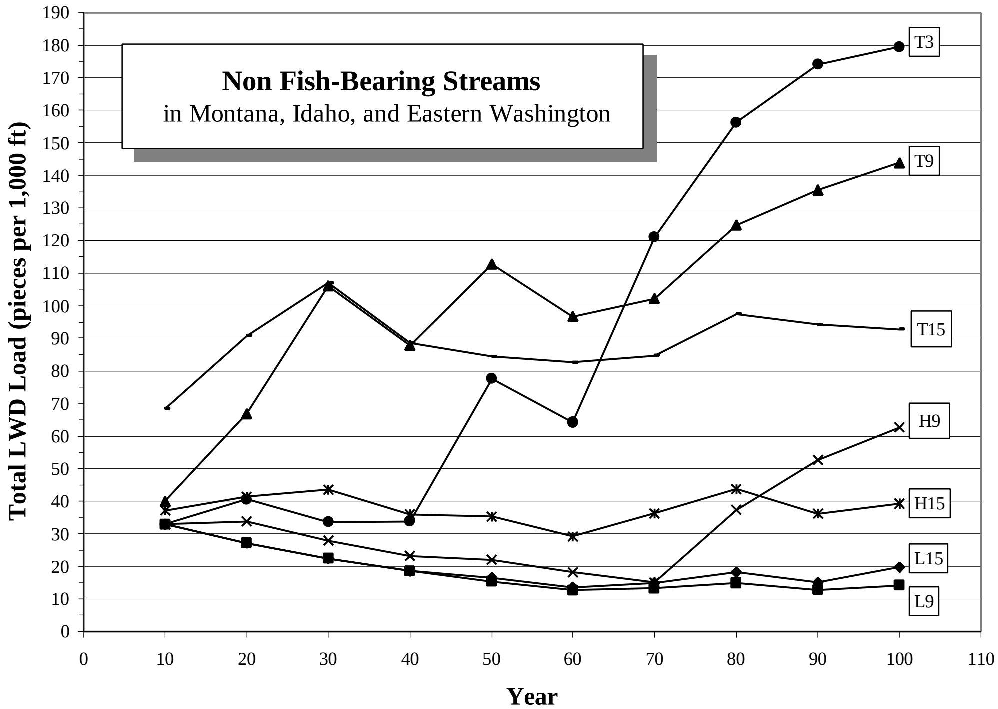
Since non fish-bearing streams are likely to be smaller and steeper than their fish-bearing counterparts, two important assumptions were altered in the modeling effort. First, simulations were only run for 5-ft channels. Second, no bank erosion-derived LWD inputs were added to the LWD estimates derived from stand-suppression mortality.

An initial finding is that a greater number of the dominant riparian stand types have sufficient numbers of trees to allow harvest than was seen for fish-bearing streams. Seven of the nine stand types offer some harvest opportunity; only the L3 and H3 stands could not be harvested immediately, owing to low densities of trees in particular size classes.

3.2.2.1 Trends in LWD Loading in Non Fish-Bearing Streams

For those stands along non fish-bearing streams that are harvestable, four out of seven show stable or increasing LWD loads over 30 years (Figure 16). Three of these are also stand types that support commercial harvest along fish-bearing streams (i.e., H15, T9, T15). The fourth is stand type T3, which after 100 years is projected to produce the highest LWD loads of any stand type under Montana's Class 2 SMZ guidelines, and rivals T9 and T15 stands under more stringent guidelines for fish-bearing (class 1) streams. The effects of thinning on residual tree growth may explain the tremendous LWD inputs from the T3 stands.LWD loads are expected to decrease for three stand types (L9, L15, H9) in response to timber harvest according to Class 2 guidelines (Figure 16).

Figure 16. Trends in LWD loads for non fish-bearing eastside streams after timber harvest in riparian stands L9, L15, H9, H15, and T3 – T15. *Harvest simulations followed guidelines of Montana's SMZ rules for non fish-bearing (Class 2) streams. Channels were assumed to be 5-ft wide and began loaded with 39 pieces of LWD per 1,000 ft.*

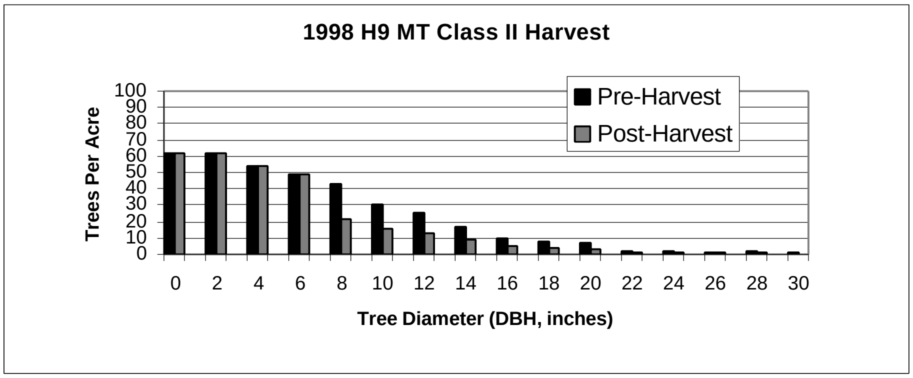


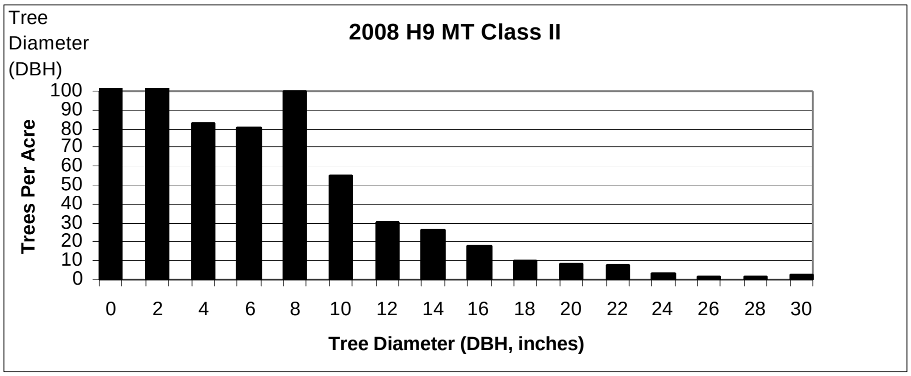
Only one stand type (T3) produced enough LWD to reach the target mean LWD load of 163 pieces per 1,000 ft of channel. We suspect that this highlights the conservative nature of the LWD modeling assumptions used in our analyses. Even with no timber harvest within 100-ft of streams, only T9 and T15 stands generated this much LWD(Figure 14). The number of qualifying LWD pieces that enter channels is no doubt much more than one piece per fallen tree. Tree limbs, stem breakage, etc., can multiply potential inputs.

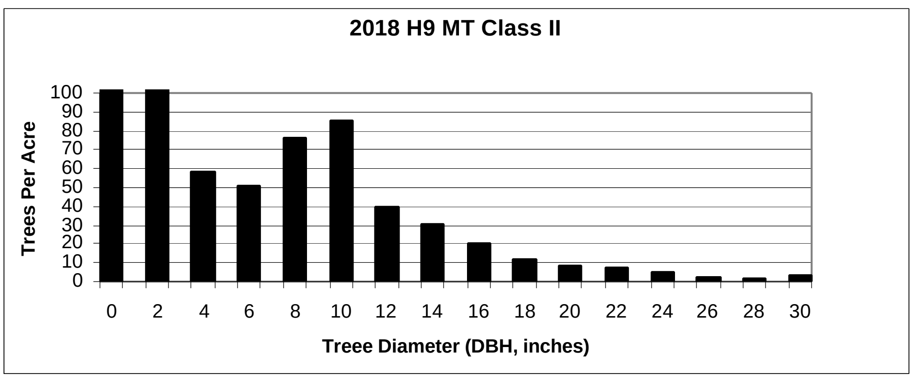
3.2.2.2 Size Distribution of Recruitable Trees

LWD piece size as well as number is important to consider. To determine the size of trees that are available for future LWD recruitment to headwater streams throughout a typical harvest cycle, the FVS model was queried. Results show that the SMZ law requirement for harvesting trees "representative of the existing diameter distribution" perpetuates a range of sizes for trees in riparian areas (Figure 17). This is further illustrated in a pre- and post-harvest visual representation of stand conditions in stand types H9 and T15 after harvest according to Montana's SMZ rules for Class 2 streams (Figures 18 and 19). The practice of leaving a representative diameter distribution should ensure that both large and small trees are available to supply LWD to non fish-bearing streams via natural recruitment processes. The need for larger diameter trees as step-forming agents in non fish-bearing channels was emphasized in the small stream assessment (Appendix B, part 2). These conclusions can also be applied to fish-bearing streams.

Figure 17. Pre- and post-harvest diameter distribution of trees in riparian stand type H9 when harvested according to Montana's rules for non fish-bearing (class 2) streams.







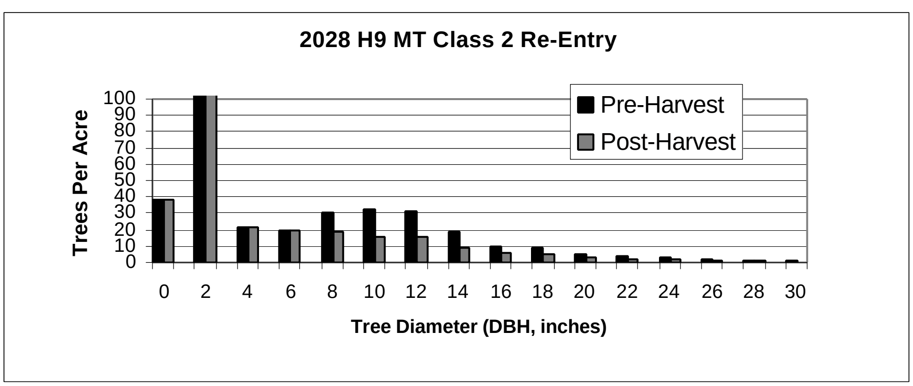


Figure 18. Visual representation of pre- and post-harvest stand conditions for riparian stand type H9 when harvested according to Montana's rules for non fish-bearing (class 2) streams.

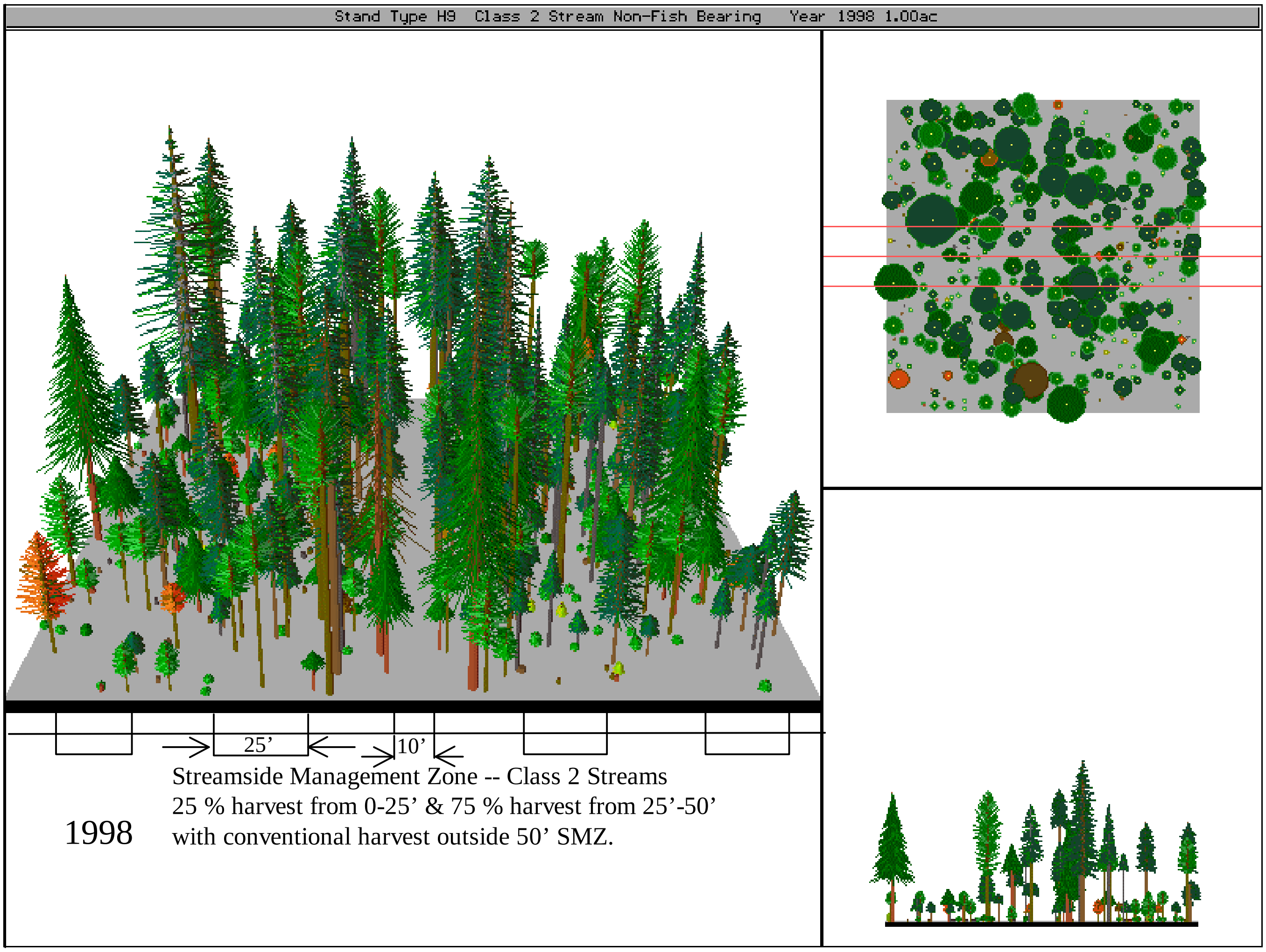
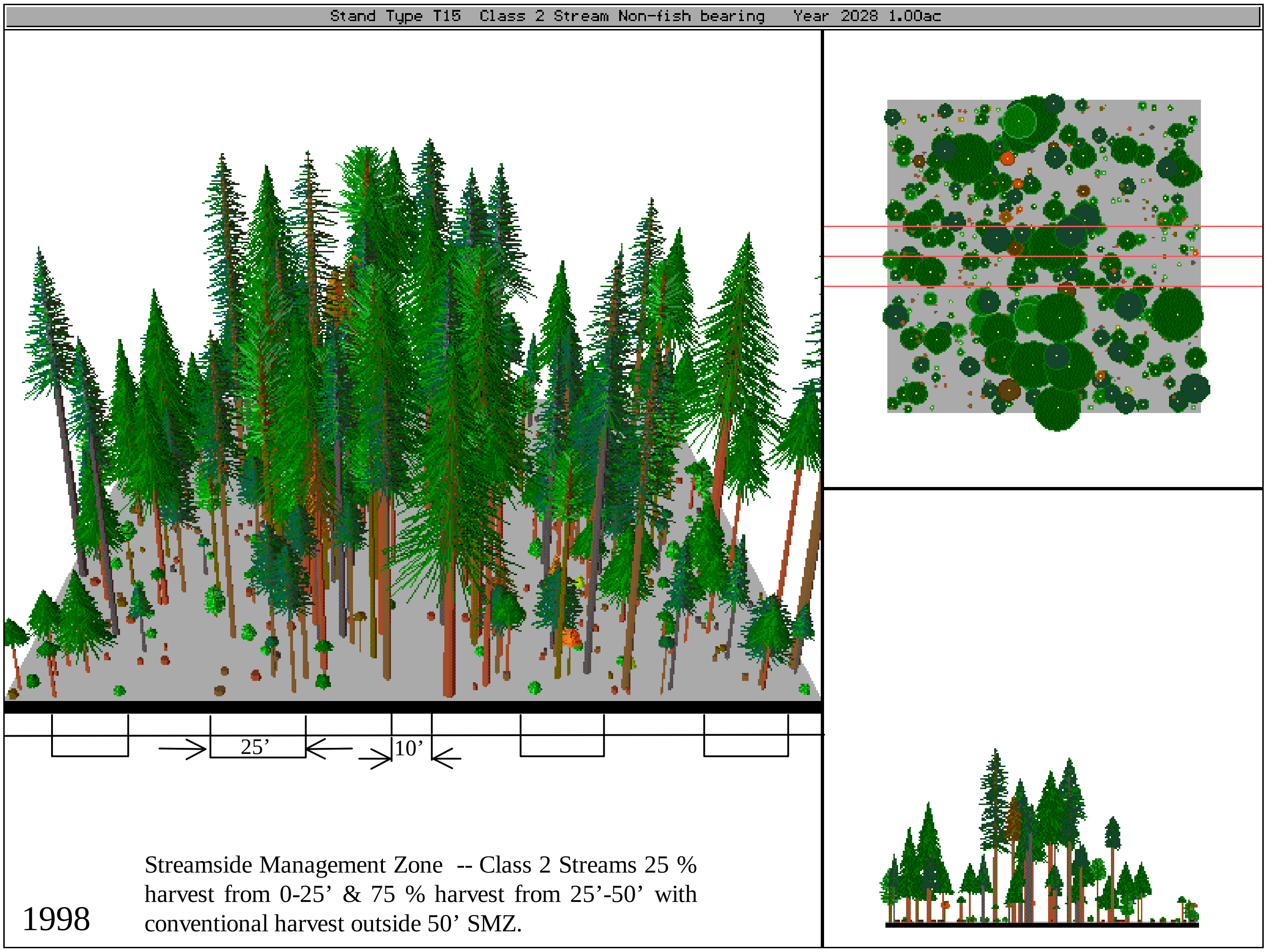


Figure 19. Visual representation of pre- and post-harvest stand conditions for riparian stand type T15 when harvested according to Montana's rules for non fish-bearing (class 2) streams.



3.2.3 Comparative Performance of Idaho's Existing Riparian Management Rules

Thus far, the focus for fish-bearing and non fish-bearing streams east of the Cascade Mt. Crest has been on comparing various riparian management strategies to Montana's current SMZ laws. This is because the bulk of bull trout-occupied streams in the project area occur in Montana. To evaluate the outcome of existing rules on LWD loads in Idaho, Idaho's leave-tree requirements were examined. No comparison was made to Washington regulations, because the riparian protection measures in that state are being revised.

Table 13. Comparison of timber harvest opportunities for dominant riparian stand types in Montana and Idaho, based on existing riparian protection regulations. *Boxes with an "X" denote current harvest opportunities.*

Stand Type	Montana		Idaho Class I streams			Idaho Class II streams
	Class I streams	Class II streams	>20 ft	10'-20'	<10 ft	
L3						
L9		X			X	X
L15		X				X
H3				X	X	X
H9	X	X	X	X	X	X
H15	X	X	X	X	X	X
T3		X				X
T9	X	X				X
T15	X	X	X	X	X	X

Montana and Idaho streamside harvest rules differ on several points. The chief difference lies in the diameter distribution of post-harvest leave trees. Montana SMZ rules require leaving 87 TPA >8" dbh or 50% of the TPA >8" dbh, whichever is greater. There are no specifications for trees <8" dbh. In Idaho, the residual tree distribution is divided into four size classes that include trees as small as 3" dbh. Minimum residual TPAs are set for each class (for a comparative summary of riparian rules and regulations for Montana, Idaho, and Washington, see Sugden and Light 1998). Also, unlike Montana, Idaho's requirements vary depending on stream size.

These differences increase the harvest opportunities in the nine dominant eastside riparian stand types (Table 13). In Idaho, additional opportunities are allowed in the H3 and T3 stand types on streams <20 ft wide. With respect to outcomes, harvest to legal limits in Idaho would be expected to produce lower LWD loads than would Montana's rules. This is because the minimum number of residual trees >8" dbh on 20-ft and wider streams is lower in Idaho than in Montana (63 and 87, respectively). The minimum residual TPA is even lower on narrower streams in Idaho. Finally, harvest of trees <8" dbh is allowed in Idaho, but not in Montana, thus further reducing the pool of recruitable trees that could serve as LWD.

Riparian harvest rules in Idaho also affect which stand types would be harvested along non fish-bearing streams. Harvest opportunities along class 2 streams are similar in Idaho and Montana

(Table 13). However, the Idaho regulations do not require residual trees >8" dbh, whereas in Montana there is a 44 TPA minimum. Consequently, harvesting under the Idaho regulations would be expected to lower LWD loading rates for larger diameter pieces.

3.2.4 Fish-Bearing Streams in Western Washington

Riparian management regulations in Washington State were being extensively revised during preparation of this document. Consequently, existing and soon-to-be-outmoded timber harvest prescriptions were not included in the performance evaluation for this area. Instead, a range of management options was simulated to examine their effect on LWD loads through time. Three simplifying assumptions were used:

1 An average channel width of 10 ft. Most of the streams flowing through Plum Creek Timber Company ownership in this portion of the planning area are smaller tributaries that occur relatively high in the drainage network.

2 Average site class = 110. This implies that growing conditions in the area will produce stands of trees where the dominant and co-dominant trees are approximately 175-ft tall ($SPTH_{200;}$ McArdle et al. 1961).

3 Half of the trees in the first 50 ft along all streams are hardwoods (principally red alder, *Alnus rubra*), but beyond this distance conifers predominate.

Table 14. Estimated in-channel LWD loads (pieces per 1,000 ft, minimum-qualifying dimensions of 10-cm diameter x 2 m length) after 30 years under different management options in streamside forests of western Washington. *For all options, channels were assumed to start with 50 pieces of LWD per 1000 ft.*

	Management Options						
Stand Type	Option 1 Clearcut to edge of channel	Option 2 0-25 ft no cut, 25-75 ft managed[2]	Option 3 0-50 ft no cut, 50-75 ft managed[2]	Option 4 0-50 ft no cut, 50-100 ft managed[2]	Option 5 0-75 ft no cut, 75-100 ft managed[2]	Option 6 0-100 ft no cut	Option 7 0-175 ft no cut[3]
WW1	27	35	35	35	35	35	35
WW2	27	78	78	78	78	78	78
WW3	27	95	102	102	102	102	102
WW4	27	95	102	102	104	104	104
WW5	27	95	102	102	104	104	104

[1]Stand descriptions:

WW1 = Immature conifer (~15 years old) to 175 ft
WW2 = Mature conifer (~50 years old) to 25 ft, immature conifer 25-175 ft
WW3 = Mature conifer to 50 ft, immature conifer 50-175 ft
WW4 = Mature conifer to 100 ft, immature conifer 100-175 ft
WW5 = Mature conifer to 175 ft

[2]Two management options apply depending on current condition of riparian forest (i.e., stand type). If stands are currently immature, then management would consist of commercial thinning to 140 TPA at age 30, with no further entry. If stands are currently mature, then management would consist of thinning to 100 TPA and no further entry. Commercial harvest (clearcutting) occurs beyond the outer edge of the riparian management zone in all options.

[3] LWD loads for a no-touch buffer of 125 ft were identical to the loads from a no-touch 175 ft buffer.

Results suggest that unlike eastside stands, after thirty years all management options besides clearcutting produce the same LWD loading outcomes (Table 14). However, when viewed over a longer timeframe (e.g., 100 years[23]), projected outcomes differ considerably (Figure 20). This is reasonable when considering the height to which conifers grow in western Washington and the time it takes for them to do this. Based on these projections, it may be more than 100 years before actions taken today will fully manifest in stream channels.

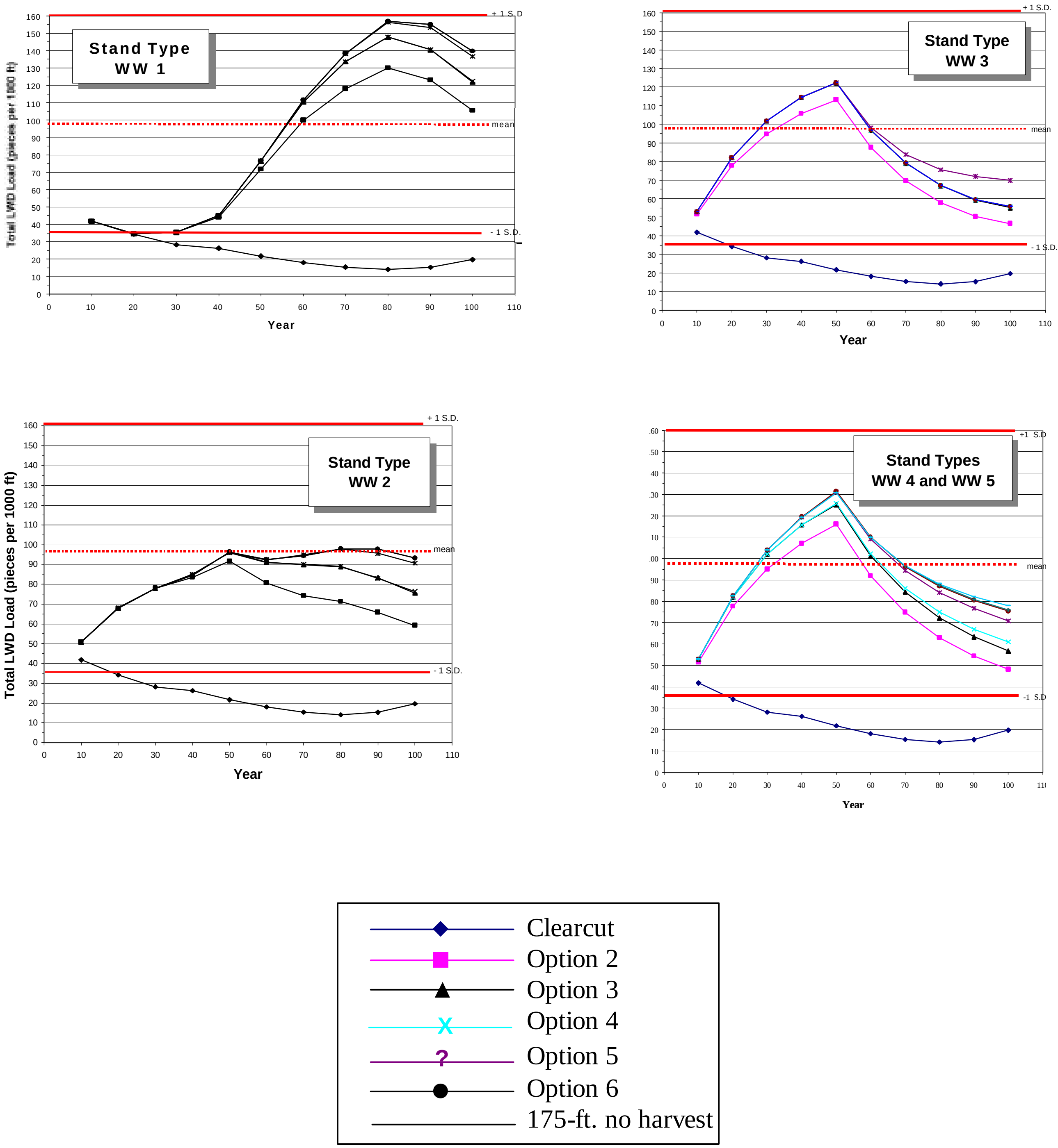
[23] We chose 100 years to examine the long-term effects of simulated harvest strategies because by this time the differences among management options have manifest. Note, however, that the reliability of growth projections using DFSIM this far into the future is questionable. One of the model's shortcomings can be seen in the continuous post-peak decline of LWD (Figure 20), which is likely produced by the lack of stand regeneration in DFSIM's simulations (i.e., young trees don't regrow to fill the holes created by the toppling of older trees).

The abundance of red alder in western Washington riparian stands affected both the timing and the amount of peak LWD loads. As sources of LWD, the importance of trees on or very near the streambank was demonstrated with the eastside assessment. The same principle holds true in western Washington, and when a substantial fraction of these trees are alder, LWD loads reflect this. In stand WW1, peak LWD loads occur at 80 years when alder has senesced and their mortality boosts inputs. For stand WW2, there is a slight bimodal peak that reflects the alder inputs after 50 years from the mature alder stands within 25 ft of the stream, followed by a second wave of inputs at 80-90 years from the alder growing in the area between 25 and 50 ft from the stream. Stand types WW3-WW5 are mature today, and therefore show a peak in LWD inputs at 50 years, again owing to the alder within 50 ft of the channel. In all stands the amount of LWD at the peak is driven by alder inputs. Conifer–dominated stands don't necessarily increase the peak loads, but they do sustain them better (analysis not shown).

In-channel LWD loads increased as the management zone width increased or as more trees were left within a given width. Highlights of the effects of different management options on LWD loads are presented below, based on results displayed in Figure 20. For reference, the mean (±S.D.) LWD loads from streams flowing through unmanaged riparian forests are shown on the accompanying figures.

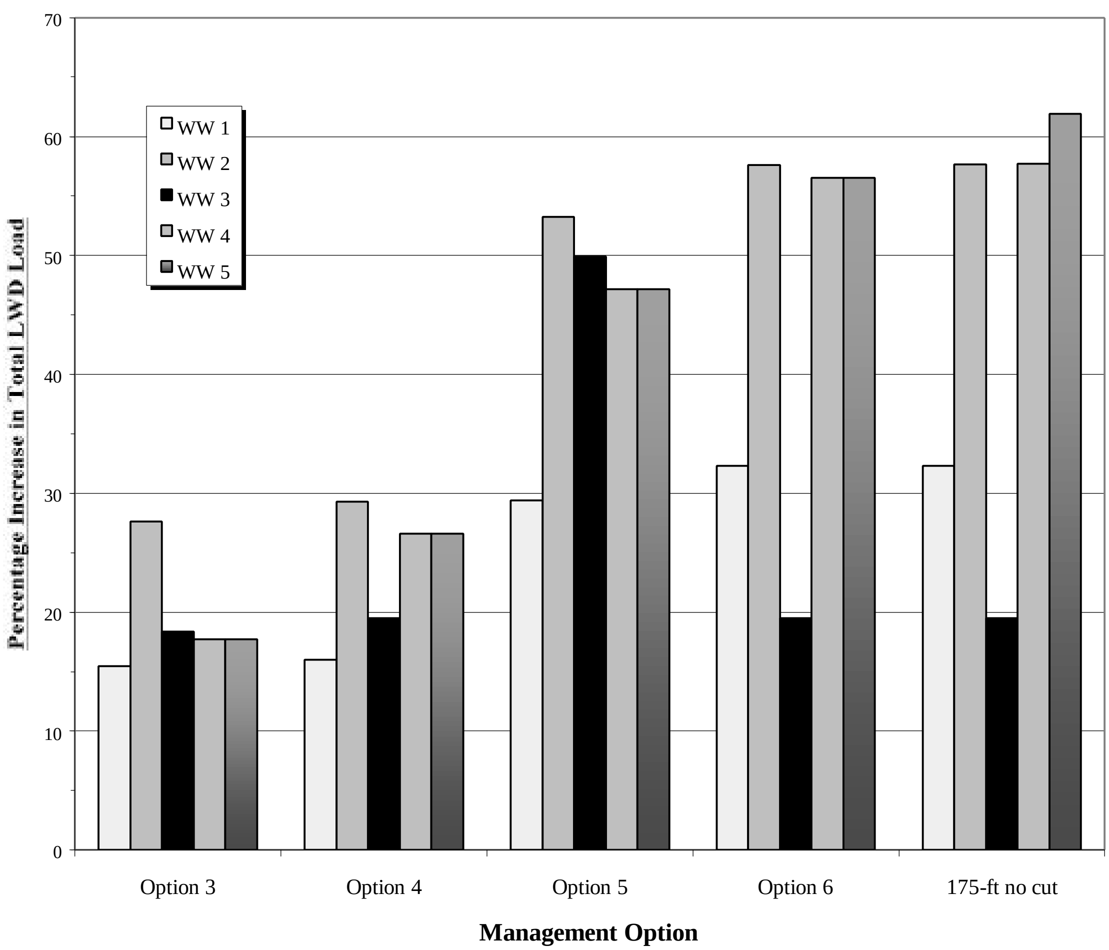
1. The effects of riparian management actions taken today may not appear in the channel for up to 100 years.
2. For stand types WW1 and WW3-WW5, all management options except clearcutting produce LWD loads that exceed the target mean temporarily, but then drop below it. The post-peak declines probably do not represent actual channel conditions but instead reflect the absence of ingrowth (a DFSIM model shortcoming). For stand type WW2, LWD loads come near to the reference mean but do not reach it, and the post-peak decline is more gradual than in other stand types.
3. The clearcutting option shows the effect of stand regeneration. Steady declines in LWD occur for 80 years, then begin to increase. However, under this management regime, LWD loads do not recover to pre-harvest levels within 100 years.
4. LWD inputs increase with increasing riparian management zone widths. No increases in LWD inputs were achieved by widening the riparian management zone beyond 125 ft for these stands (site class III, $SPTH_{200}$ ~175 ft).
5. The chosen thinning regime does not appear to greatly accelerate timing of inputs or improve loading levels. However, the effect of thinning is not fully explored in these simulations. A more elaborate analysis of thinning regimes may help optimize LWD loads through combinations of thinning and no cut areas.

Figure 20. Estimated trends in total woody debris loads in fish-bearing streams of western Washington by stand type. *LWD loads represent pieces per 1,000 ft. of stream channel with minimum qualifying dimensions of 10 cm. diameter x 2 m. length. Reference mean ± SD represents data from surveys of LWD loads in streams flowing through unmanaged riparian forests of the coastal Pacific Northwest (Bilby and Ward 1989). See text for descriptions of stand types and management options.*



An examination of LWD loads produced after 100 years provides some insight into the long-term consequences of different management strategies (Figure 21). By this time, differences among modeled outcomes have been revealed (see Figure 20). Differences are calculated relative to option 2 (a 75-ft SMZ with a 25-ft no harvest core area). The width of no-cut areas produced the greatest differences among management options. Widening the no-harvest area from 25 ft (option 2) to 50-ft (option 3) increased LWD loads by 15-28% depending on stand type. Expanding the managed zone from 75-ft to 100-ft (options 3 vs. 4) had little effect on stands with a large proportion of immature trees in the first 100 ft (i.e., WW1-WW3), but produced about 10% more LWD for stands WW4 and WW5. The differences between 100-ft and 175-ft no-harvest SMZs (options 6 vs. 7) were not remarkable.

Figure 21. Increases in LWD loads for western Washington fish-bearing streams after 100 years, relative to management option 2. *See text for descriptions of stand types and management options.*



3.2.4.1 Effect of Bank Erosion on LWD Loads in Streams of Western Washington

In contrast to findings for eastside streams, bank erosion inputs were relatively minor for most westside streams. After 30 years, the estimated bank-erosion inputs (from Table 9) increased LWD loads by approximately 11% for stand types WW2 – WW5. Bank erosion would contribute proportionately more of the LWD load at 30 years for stand type WW 1, where LWD loads were boosted 43% by bank erosion sources. Because these estimates apply to the time period when LWD inputs from stand suppression mortality were near their lowest point in the 100-year modeling window (Figure 20), the relative importance of bank erosion inputs would be expected to diminish further as time progressed.

3.2.5 Non Fish-Bearing Streams in Western Washington

The evaluation of timber harvest effects on LWD loads in perennial, non fish-bearing streams of western Washington was limited to two management options: a 50-ft SMZ with no timber harvest and clearcutting to the stream bank. These were selected based on a combination of factors. First, managers provided input on cost constraints associated with wider management zones for this type of stream in the planning area. Second, a buffer of 50-ft is generally regarded as adequate to meet the stream shading requirements and maintain water temperatures within state water quality standards. Third, the long-term functional integrity of narrower SMZs may be threatened by windthrow. Fourth, a 50-ft buffer width is typically prescribed as adequate to protect unstable slopes such as inner gorges that border many of these small streams.

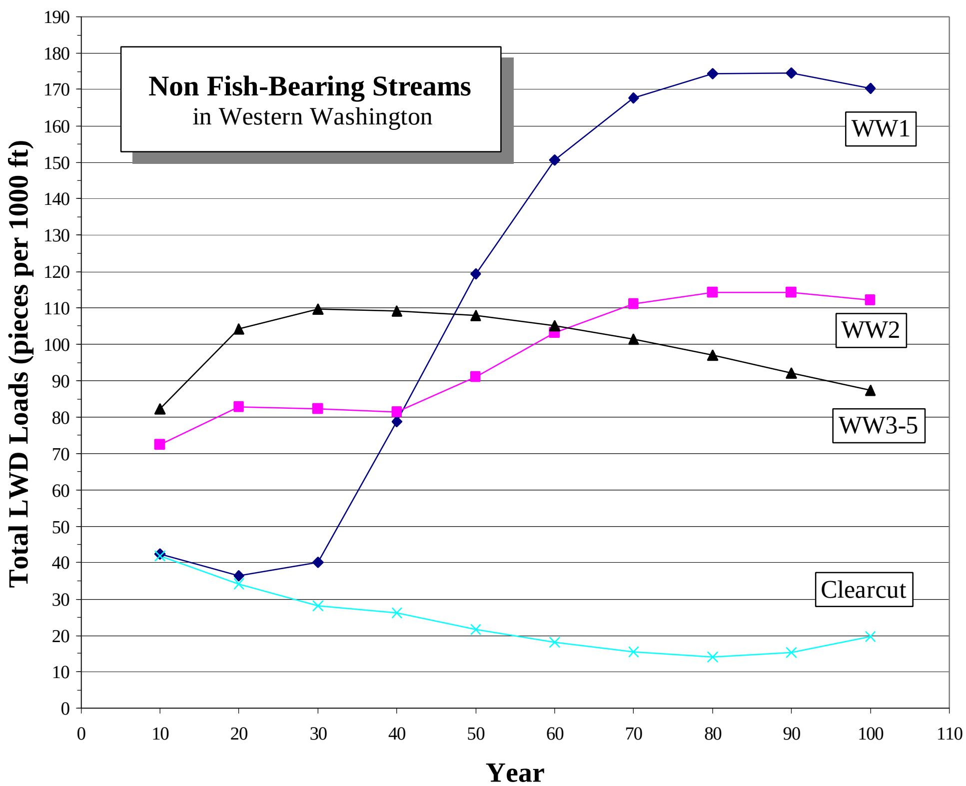
Clearcutting simulations were used to illustrate the effects of recent forest practices on LWD loads, as well as to provide insight into the LWD loads that would be generated for reaches where yarding corridors remove streamside trees. The average channel width was assumed to be two feet. A target of 163 LWD pieces per 1,000 ft was again used to evaluate performance of these management options (see section 3.1.1, Setting Targets).

With a 50-ft SMZ, all five stand types generated LWD loads that were comparable to loads from wider SMZs on larger streams (Figure 22). This is undoubtedly due to the lower depletion rates for LWD in these small channels. Despite the relatively high LWD loads, only stand type WW 1 approached the target mean. As mentioned for the eastside evaluation, we believe this is either a result of the model's underestimation of LWD inputs, or an unrealistic target, or both.

Clearcutting led to declines in LWD, but not indefinitely. By 80 years the LWD inputs from regenerating stands exceeded outputs, and thereafter LWD loads began to replenish.

These results indicate that 50-ft no-harvest SMZs will increase or maintain existing levels of in-channel LWD in non fish-bearing perennial streams. In so doing, LWD functions in these channels will be maintained. The results also suggest that if catastrophic disturbances such as debris flows removed all LWD from channels bordered by these SMZs, then LWD replenishment would begin almost immediately. In places where yarding corridors remove trees along significantly long sections of stream (i.e., beyond the range of LWD inputs from adjacent reaches with SMZs), LWD loads will temporarily be reduced. These 50-ft SMZs also provide a full range of LWD sizes; very large pieces including trees with attached rootwads are completely provided for, as are other sizes down to and below the minimum qualifying LWD piece size. If SMZs of this width can maintain other riparian functions to the degree necessary to support fish habitat downstream, then managers should be able to operate profitably in the headwaters of western Washington without compromising key ecological functions provided by riparian areas. A review of other functions for all portions of the channel network is presented in the Discussion.

Figure 22. Trends in LWD loads for non fish-bearing streams in western Washington after timber harvest in riparian stands WW1–5. *Channels were assumed to be 2 feet wide and began loaded with 50 pieces of LWD per 1,000 ft.*



4.0 DISCUSSION

The goal of this work is to provide the technical means and rationale for selecting riparian management options for Plum Creek's Native Fish HCP. To achieve a balance between fish habitat protection and riparian timber harvest, we sought a means of discerning where stream channels were most vulnerable to timber harvest. A review of the scientific literature and an analysis of local field data led to the conclusion that sensitivity could be described using a combination of channel gradient, fish presence, and the life history stage(s) supported by a particular stream reach. Other situations of special concern, nested within the general gradient classes, are channel migration zones (Appendix A).

Next, we gathered riparian stand information from the eastside and westside portions of the project area, and used this to describe the current condition of dominant riparian stand types. There were nine types in eastside riparian forests, with stand density and tree size being their most distinguishing features. There were five types in western Washington, distinguished mainly by the mix of age classes. For a range of potential management options, the LWD loads produced by these stands were then simulated using a combination of the FVS growth and yield model (eastside) DFSIM (westside), the RAIS LWD loading model, and bank erosion estimates. These steps generated the information needed to assist managers in selecting riparian harvest strategies that appropriately match local resource sensitivities (i.e., type-specific prescriptions, or strategy "C" in Figure 1), per the process displayed in Figure 7.

4.1 Tailoring Management Actions to Address Fish Habitat Vulnerabilities

Results of the management simulations suggest that for streams east of the Cascade Mountain Crest, Montana's existing riparian protection measures will provide LWD loads in fish-bearing streams that are well within the range of natural variability. This range is wide, however, and makes for a large target. Furthermore, the LWD loads at the lower end of this range are not necessarily sufficient to maintain high quality habitat. These findings suggest that Montana's existing management system provides a substantial baseline of conservation, but improvements are possible. The fundamental elements of Montana's guidelines can also be considered as a baseline of conservation measures in Idaho and eastern Washington. Though Idaho's existing SPZ rules were only partly evaluated, the results suggested that overall they would perform less well for providing LWD than Montana's SMZ guidelines.

In searching for situations where riparian conservation measures can be strengthened, managers face the challenge of crafting strategies that reduce risk more where vulnerability is greatest. How can this be done? In our view, narrowing the target LWD loads closer to the "bull's eye" of the mean is a reasonable approach to risk reduction. In other words, selecting a management option that produces LWD loads that are within the 95% confidence interval around the mean is less likely to result in adverse impacts to habitat than an option that produces LWD loads within a standard deviation of the mean. For example, LWD loads as low as 30 pieces per 1,000 ft are produced naturally in some areas and would fall within one standard deviation of the mean. Selecting a riparian strategy that only generates 30 pieces/1,000 ft where a stand could produce more LWD, however, would not be appropriate for a plane-bed/forced pool riffle reach (1.5-3% gradient) in a Tier I watershed (i.e., the channel type and location with the highest sensitivity to LWD loss). Other confidence intervals (e.g., 99.9%) could be used to set lower target thresholds and thereby address the range of sensitivities described in Table 5. Defining reference mean LWD loads for each of the stand types would greatly facilitate implementation of this approach.

The reference mean we used to evaluate management options was useful in demonstrating that no existing or contemplated management options result in severe declines in LWD loads, but was less useful in evaluating the relative effect of the modeled options within a given stand type. This is due to the composite nature of the reference mean. We do not know which stand types bordered the stream reaches in the studies we used to calculate the reference mean. The wide variability around the average wood loading values in eastside (78±57 S.D. pieces/1000 ft) and westside (98±62 S.D. pieces/1000 ft) streams suggests a wide range of stand types were sampled. Ideally, to account for innate differences in LWD loads among stands, the mean LWD load and natural variability would be defined for each stand type. This would allow establishment of stand-specific management targets and would aid in measuring the effectiveness of different management options. This will be possible as the HCP is implemented and monitored.

Riparian forest stand characteristics greatly influenced the LWD loading potential, and therefore the potential habitat conditions that a given stream reach can provide. There was a ten-fold difference in modeled LWD loads between stands at opposite ends of the tree size and density spectrum (L3 and T15, respectively; Table 11). Incorporating stand type information into decision-making processes may significantly improve a manager's ability to refine local riparian harvest plans. However, to a significant degree the inherent capacities of different stands to provide LWD and other riparian functions are already accounted for in Montana's existing riparian management guidelines. The SMZ guidelines for fish-bearing streams contain minimum leave-tree requirements that significantly reduce timber harvest in riparian stands with low tree densities and small tree sizes (i.e., stand types L3-L15, H3, T3). These stands would naturally be the least able to provide LWD and other functions that support stream ecosystems. Conversely, stands with very high densities of large trees (i.e., T9, T15) are still capable of generating relatively high LWD loads and canopy closure levels even after removal of proportionately more trees under these same guidelines.

The riparian functions provided by stands with intermediate tree densities and sizes (i.e., H9, H15) may not be as well protected. These stand types have the potential to generate LWD loads at levels near to the reference mean, but harvest to legal limits of Montana's SMZ guidelines results in loads that are well below average (Table 11, Figures 14-15). This effect is more pronounced for H9 stands, and for larger channels. We did not attempt to translate the reference mean LWD frequency into pool frequency or other habitat variables and therefore we did not explicitly quantify the expected habitat benefits of LWD loads attained by any management option. However, the degree of LWD reduction suggests that managers may want to consider providing additional protection to these stands, or at a minimum be certain they are monitored carefully in the adaptive management process. The response of channels and fish habitat to harvest in these stand types is important to understand because the H9 and H15 stands comprise approximately 27% of the fish-bearing stream miles in the project area. Of all the stream reaches in the project area, decisions made here may have some of the largest ecologic and economic consequences.

Within H9 and H15 stands, or among stands generally, there are two situations where additional conservation measures may be warranted as insurance against potential impacts of timber harvest on aquatic ecosystems. Earlier in this report we identified plane-bed/forced pool-riffle channels (1.5-3% gradient), and channel migration zones (typically 1.5% to 6% gradient, see Appendix A) as being particularly sensitive to loss of LWD. Increased protection in the form of increased leave trees in the SMZ and CMZ could provide insurance against unforeseen loss of LWD from streamside timber harvest. Because of the presence of bull trout, Tier 1 drainages would be first priority for these additional conservation measures (Figure 6).

Western Washington riparian conditions differ significantly from those elsewhere in the project area, and therefore require different management options to achieve the objectives of the HCP. Trees grow much taller in western Washington, which means that LWD can reach streams from farther away, and therefore it takes longer for trees at distance to grow within range of the stream. This requires a longer timeframe for examining projections, otherwise all management options appeared to produce the same near-term result (e.g., after 30-years). Viewed for 100 years or more, actions taken today can markedly affect future in-channel conditions.

All simulated management options for western Washington performed well against the reference mean except for stand type WW 2. Option 2 (no harvest within 25-ft of the channel plus 50-ft managed) consistently underperformed relative to other options. Additionally, the 25-ft no harvest portion of this option may not provide adequate canopy closure (see “review of functions” section below). Because of the even-aged management of westside stands, adjusting the width of the no-harvest area next to the stream may be the most effective way to affect LWD loads and thereby distribute conservation measures among the channel and fish habitat sensitivities described in Figure 6. Thinning trees at distances beyond 50 ft may also accelerate the rate at which maturing trees grow within range of the stream.

The presence of red alder immediately adjacent to most western Washington streams affects both the amount and timing of LWD loads through time. Alder and other hardwoods are a natural and important tree species along stream courses throughout the project area. To the extent that the current abundance of these species in riparian areas has been influenced by past management activities, there may be opportunities for ecological restoration (see “opportunities for stream ecosystem improvements” section below).

For non fish-bearing streams, Montana's existing SMZ guidelines were judged to be adequate to support a properly functioning aquatic ecosystem in the interior Columbia River Basin, and the simulated 50-ft SMZ in western Washington was likewise judged to be adequate. Both LWD and canopy closure were maintained at high levels with these options, even though these management options did not generate sufficient LWD to meet the proposed target, based on RAIS simulations.

Evaluation of the adequacy of the proposed riparian prescriptions for non-fish bearing streams must take into account several factors. These include uncertainty regarding target levels of LWD, long-term trends of LWD load predicted by RAIS, and scientific uncertainty regarding definition of adequate ecological function of LWD in non-fish bearing streams, particularly with respect to indirect effects on downstream fish habitat.

The target level selected for LWD in non-fish bearing streams was based on available data, a key element of which (the ratio of LWD pieces to LWD steps), was drawn from a relatively small data set developed from a field study in northwest Washington. Other data on LWD steps from other regions strongly suggest there are few differences among regions and that the small, unique data set from northwest Washington may be expected to be reasonably representative. The LWD frequency: LWD step ratio was applied to other data drawn from a wider geographic region on the frequency of LWD per unit channel length to generate the LWD reference target of 163 pieces per 1,000 ft. The strength of this simple approach to setting a target is that it is based on field data, and it targets a critical component of LWD function in these streams: sediment storage and channel grade control by LWD steps. Its weakness is that it relies on a limited data set from a region that has significant climatic and ecological differences from most of the project area. Because these data were from a region with relatively wet, warm climate and commensurate high growth rates, it is reasonable to suppose that these data represent conditions of high LWD recruitment and loading. Consequently, the resulting target is likely to be conservative in that it sets goals that represent the LWD-rich end of the spectrum of LWD loading in small streams.

The simulation model of LWD recruitment (RAIS) is quite conservative in its estimates of LWD recruitment to non fish-bearing channels. In these steeper, smaller streams, the model does not attempt to account for significant LWD inputs from branches, twigs, brush, the boles of fallen trees and their fragments, or mass wasting. Hence, the estimates of LWD recruitment and loading generated by RAIS are likely to substantially underestimate actual levels.

Finally, what is known about the effects of LWD in non-fish bearing headwaters on downstream fish bearing waters is limited. Quantitative studies of LWD steps, sediment storage and sediment transport (e.g. Megahan 1982, O'Connor and Harr 1994), suggest that the sudden or more gradual (over a period of decades) loss of LWD and LWD steps from headwater streams can release quantities of sediment that could be sufficient to negatively affect downstream fish habitat. Even so, the impact on downstream habitat is not predictable, and would depend on a variety of watershed factors. Hence, even "worst case scenarios" are not conclusively detrimental to downstream habitat. Moreover, the results of these studies imply that if LWD recruitment of a variety of size classes is maintained over time, even at a reduced but substantial rate, it is unlikely that downstream sediment impacts on fish bearing streams would be either measurable or significant.

RAIS simulations suggest that the proposed riparian prescriptions would generate an increasing trend in LWD recruitment for most stand types, with gradual decreases in streams adjacent to low density riparian stands in eastside forests over periods of decades. Taken at face value, this outcome indicates some cumulative balance between gains and losses from the current condition distributed over space and time, but with more gains than losses. If one considers that LWD inputs to streams may well be dominated by disturbance to riparian forest stands (e.g. wildfire, windstorms, insect infestations) that are not well-represented in RAIS simulations, even those stream types predicted to have a declining long term trend may in fact prove to have higher recruitment rates in the long term.

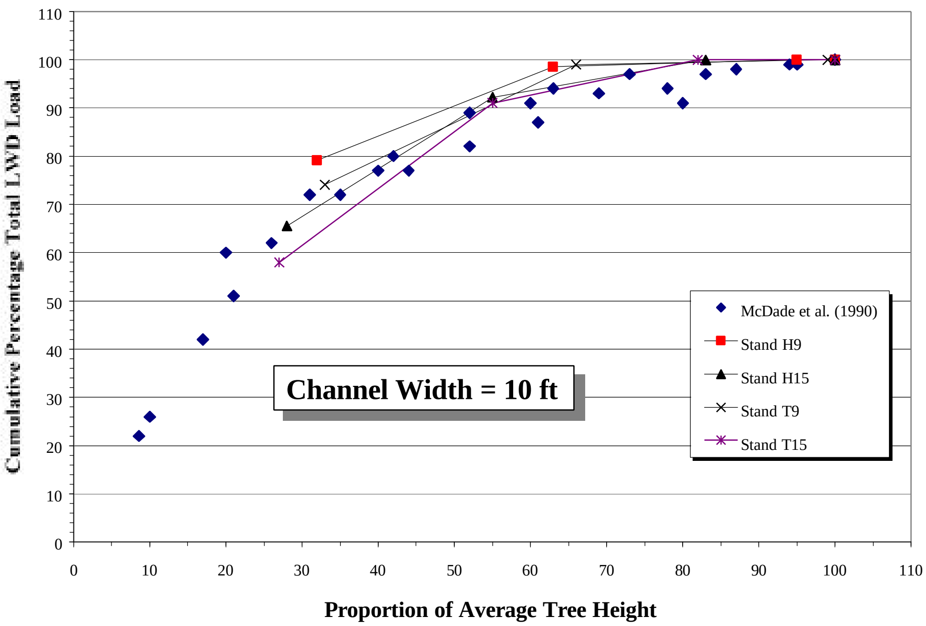
Despite predictions that LWD recruitment will not meet the LWD target, it is quite likely that LWD functions will be maintained. Estimates of recruitment are conservative, and the high target is likely to be at the high end of the LWD loading spectrum. Baseline riparian protection measures such as those currently being implemented in Montana and those that were simulated for western Washington, will maintain a long-term source of LWD inputs to streams. These will in all likelihood be sufficient to provide properly functioning LWD for the aquatic ecosystem. As with other conclusions of this report, these regarding LWD loads in non fish-bearing streams can be verified through adaptive management.

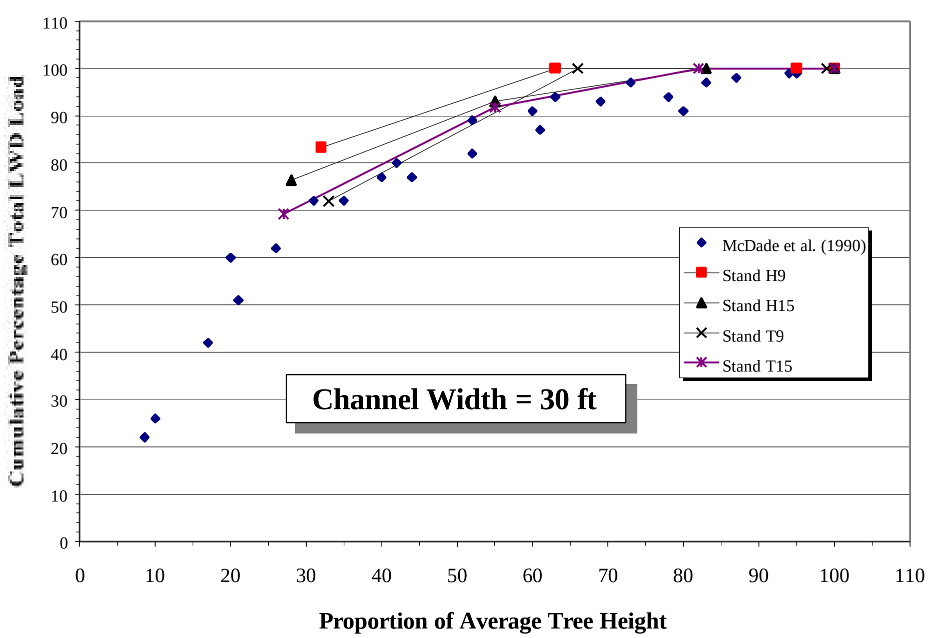
4.2 Empirical Validation of Modeled LWD Loads

In an attempt to verify that the various models used in our analyses were working together to generate realistic results, we compared modeled with measured relationships between LWD recruitment and source distance. McDade et al. (1990) described this relationship for unmanaged stands of conifers and hardwoods in western Oregon and Washington. They found a curvilinear function of diminishing LWD inputs with increasing distance from the stream. The equations were different for hardwoods and conifers, and also for mature vs. old-growth conifers. However, when distance from the channel is expressed as a proportion of average tree height, the differences among tree species and stand ages disappear. We used the normalized data from McDade et al.'s empirical work to compare with the modeled outputs from RAIS for unmanaged stands.

Results show that the LWD source distance relationships generated by the RAIS model are comparable with those measured by McDade et al. (1990). For eastside stand types H9 and T9 the RAIS model overestimated the relative contribution of near-stream trees (Figure 23). The differences were less than ten percent. To translate the differences into on-the-ground distances, the McDade et al. results showed 85% of LWD inputs originated within a distance equal to 50% of the average height of trees in the stand. Modeled results showed 85%-94% of inputs came from within the same 50% of the average tree height, depending on stand type and channel width. Results for smaller (10-ft) channels were closer to McDade et al. than larger (30-ft) channels.

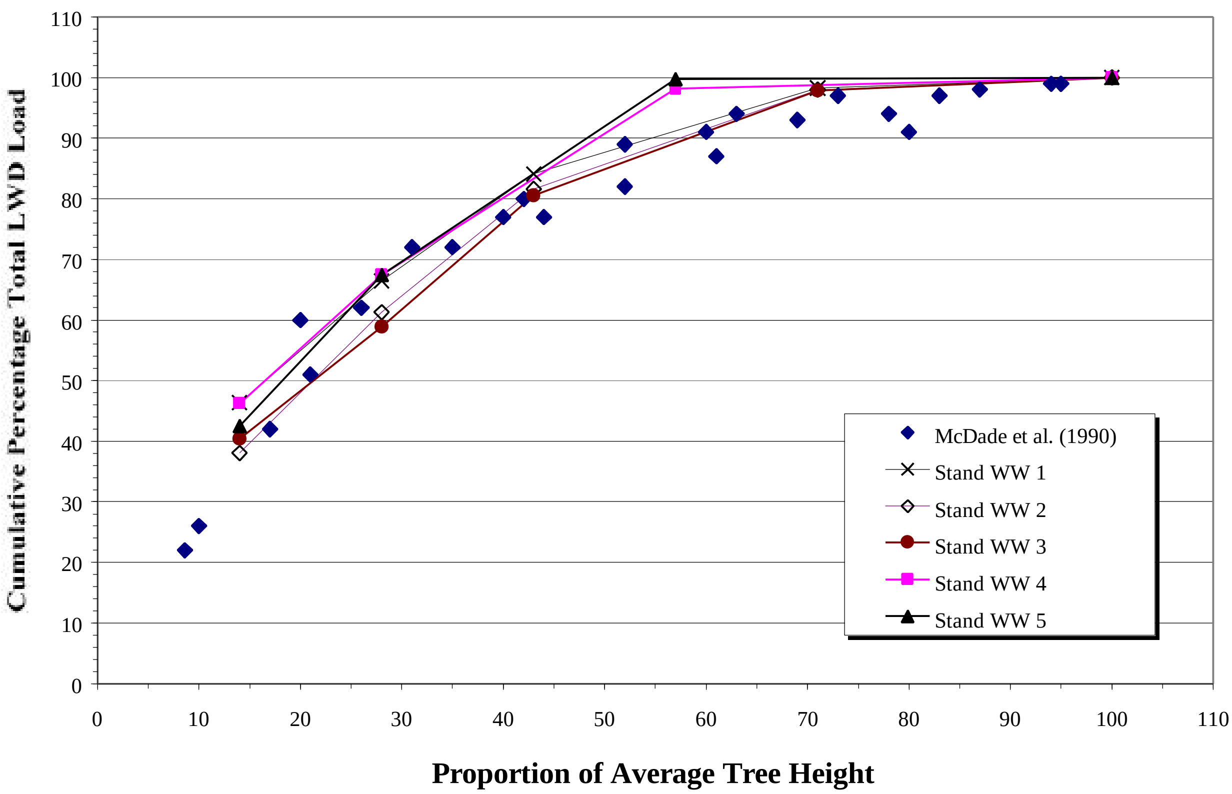
Figure 23. Comparison of modeled (RAIS) vs. empirical (McDade et al. 1990) relationships between source distance and proportion of LWD inputs for riparian stands in the Interior Columbia River Basin. *Source distances were scaled to the proportion of average tree heights in each stand type. See text for a discussion of the RAIS model (Welty 1998) and assumptions.*





The LWD source distance function for modeled westside stands was similar to that for eastside stands (Figure 24). Again the RAIS model overestimated near-stream inputs, but the differences were less than 5% for stands WW1-WW3, and 10% or less for stands WW4 and WW5. Also, the differences only appeared in the last 15% of LWD inputs. Because the RAIS model relied on input from the two forest growth simulators (FVS and DFSIM), this comparison of modeled outputs vs. empirical data provides some confidence that the models are working together to fairly accurately simulate the dynamics of riparian forests with respect to LWD recruitment. Whether the RAIS outputs accurately predict the actual amount of LWD present in the channel through time is a topic that will be discussed in the next section.

Figure 24. Comparison of modeled (RAIS) vs. empirical (McDade et al. 1990) relationships between source distance and proportion of LWD inputs for riparian stands in western Washington. *Source distances were scaled to the proportion of average tree heights in each stand type. See text for a discussion of the RAIS model (Welty 1998) and assumptions.*



4.3 Modeled vs. Actual LWD Loads

As with any modeling effort, a valid question is: How well do the modeled outcomes reflect reality? The answer to this question partly depends on how estimated outcomes, in our case LWD loads, are to be used. For instance, it would be naive to expect to find exactly the modeled number of LWD pieces per 1000 ft in a given stream channel, but on average the estimates appear reasonable. Estimated LWD loads were in most cases within published natural ranges, and results conform with the expected relationships between riparian forest condition and in-channel LWD loads. That is, lower density stands (e.g., L3-L15) generated less LWD than more heavily stocked stands (e.g., T3-T15). Several researchers who examined both stand conditions and in-channel LWD characteristics observed similar relationships (Hayes 1996, Knight 1990, Bilby and Wasserman 1989). Therefore, for the purpose of evaluating riparian management options, the estimated LWD loads are considered reasonable. Further work on the relationships between stand type and LWD loads would help confirm the accuracy of modeled LWD loads by stand type.

4.4 Treatment of Uncertainty

An important step in any successful planning process is an explicit consideration of risk. Risk in this context is a direct function of the cumulative effects of critical uncertainties associated with recommended treatments (Lichatowich et al. 1995). Throughout this analysis, many assumptions were made to address scientific uncertainties. Some of these concerned the strength of linkages between elements of the riparian cause-effect pathway (Figure 4), that led to prioritization of riparian protection levels (Figure 6). These were needed to guide management objectives toward achievement of the overall goal: to find a credible balance between competing resource uses in riparian areas.

These risks are lessened, in large part, by knowing the management prescriptions developed from the analysis would not be applied along all stream reaches in the project area over the planning time frame. This is a result of both spatial and temporal management constraints. For example, if Montana's existing riparian management rules were applied throughout a bull trout drainage, then the analysis shows that in most instances the resulting LWD loads would be maintained within natural ranges. However, land ownership and management strategies are intermingled in the project area, and rarely will any given set of management options be applied to an entire drainage. More than half of the total stream miles in the planning area flow through lands owned by the U.S. Forest Service (see Sugden and Light 1998). These lands are anticipated to have no-harvest riparian buffers that are many times wider than this quantitative analysis showed would be needed for maintaining inputs of LWD. The end result is reduced risk to bull trout habitat from uncertainties in estimated outcomes.

From a temporal perspective, some riparian areas may not be entered during the plan term, if ever, owing to the combined effect of leave-tree constraints in existing rules and inherent stand potential. And, as discussed in the Conceptual Framework section, not all stream reaches share the same level of sensitivity to wood gain or loss. The most sensitive category of stream channel (1.5-3% gradient) amounts to approximately 18% of perennial fish-bearing streams on Plum Creek lands in the project area. Therefore the pace and scale of application of riparian conservation measures afford ample opportunity to learn from adaptive management (see Hicks et al. 1999).

Other uncertainties were addressed by conservatively selecting simplifying assumptions in the modeling portions of this analysis. All major assumptions were based on published literature, and only reasonable values of model parameters were used. An estimation of the implication (i.e., conservative vs. liberal) of major assumptions is provided in Table 15.

An additional cushion against potential errors in the analysis is the influence of trees occurring in timber stands beyond the riparian area. The partial harvest methods used over most of Plum Creek's lands in the project area ensure that trees remain beyond the 50-ft. riparian management area between successive harvests. These trees vary in size and species, and provide a windthrow buffer and a well-developed forest structure for accommodating any adjustments that result from adaptive management. The character of the typical forest beyond the riparian area after conventional harvest is illustrated in Figures 25 - 28 for stand types H9, H15, T9, and T15, respectively.

Together, these conditions greatly improve the likelihood our forecasts are correct. Remaining uncertainties can be addressed through adaptive management.

Table 15. Descriptions and implications of major simplifying assumptions used in modeling LWD loads.

Assumption	Implication
Uniform probability for direction of tree fall (e.g., independent of hillslope gradient). (Van Sickle and Gregory, 1989)	*Conservative* (under-estimates LWD loads) Recent field studies in western Washington and Oregon demonstrate some effect of slope on tree fall direction (Andrus 1998, Minor 1997)
Initial in-channel LWD load set at one half of the mean LWD loads found in unmanaged streams (Bilby and Wasserman 1989, Huntington 1995, and Plum Creek Timber Company [unpublished data] document no significant differences in LWD loads between managed and unmanaged streams in the Interior Col. R. Basin. However, Bilby and Ward [1989] and others have found lower LWD abundance in managed streams in western Washington)	*Neutral* (May under-estimate current LWD loads in streams of Montana, Idaho, and eastern Washington, but may over-estimate current loads in western Washington)
Rustagi and Loveless (1991) tree taper equation accurately estimates taper of trees in portions of the project area east of the Cascade Mt. Crest.	*Conservative* The Rustagi and Loveless (1991) taper function was developed for trees west of the Cascade Mt. Crest. This function produces a more pronounced taper compared to taper functions developed for trees east of the crest (e.g., Kozak 1994). The effect is to place the effective height lower in the tree, therefore reducing the distance from the channel bank that a given tree could actually contribute LWD. Hence the model is conservative in the count of trees available for possible recruitment.
FVS -modeled tree mortality and bank erosion were the only LWD recruitment processes that were included.	*Conservative* Fire, windthrow, mass wasting, upstream sources, or other catastrophic input of LWD would increase LWD loading. (Bragg and Kershner 1997)
Only one piece of LWD is derived from each tree that falls	*Conservative* Breakage and branches multiply the potential LWD pieces.
LWD depletion rates derived from studies in S.E. Alaska (Murphy and Koski 1989), and coastal Washington (Grette 1985) are applicable to the project area.	*Conservative* Relatively low depletion rates may be likely for LWD in interior streams, as evidenced by the lack of a relationship between channel size and LWD characteristics.

Figure 25. Visual representation of riparian stand type H9, showing the character of the riparian forest after harvest according to Montana's SMZ guidelines for fish-bearing (Class 1) streams, and the character of the adjacent forest after conventional single tree selection harvest.

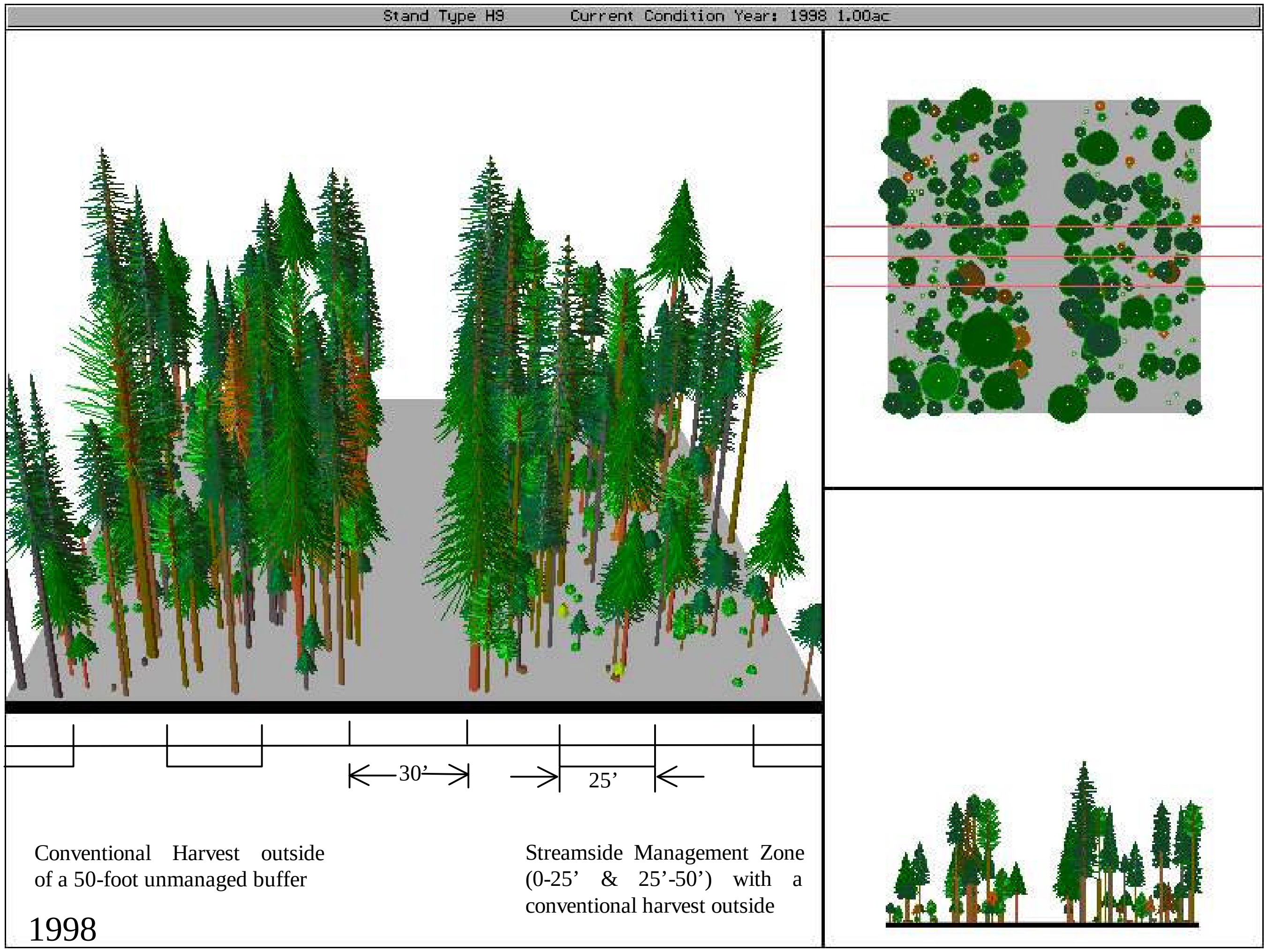


Figure 26. Visual representation of riparian stand type H15, showing the character of the riparian forest after harvest according to Montana's SMZ guidelines for fish-bearing (Class 1) streams, and the character of the adjacent forest after conventional single tree selection harvest.

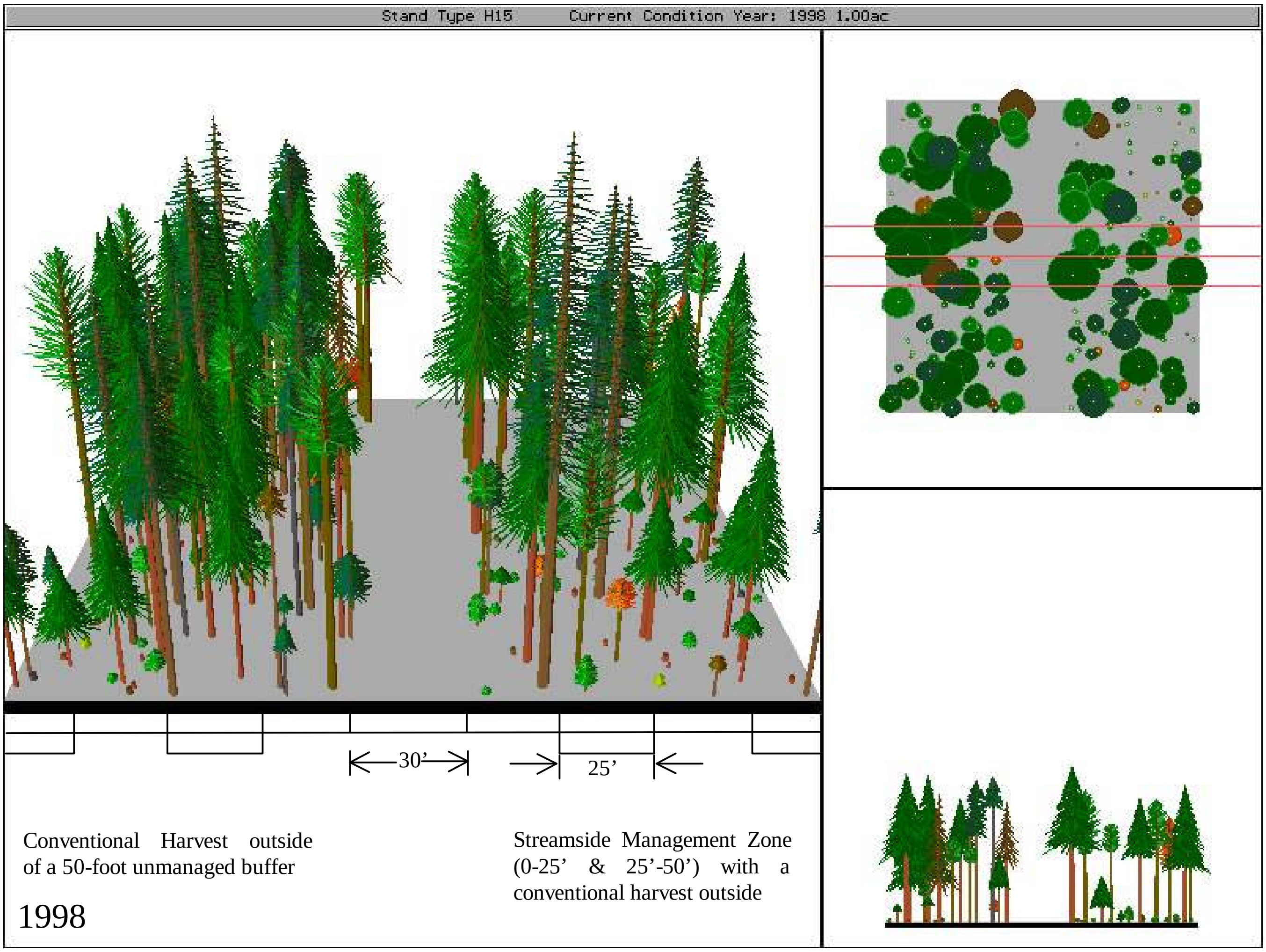


Figure 27. Visual representation of riparian stand type T9, showing the character of the riparian forest after harvest according to Montana's SMZ guidelines for fish-bearing (Class 1) streams, and the character of the adjacent forest after conventional single tree selection harvest.

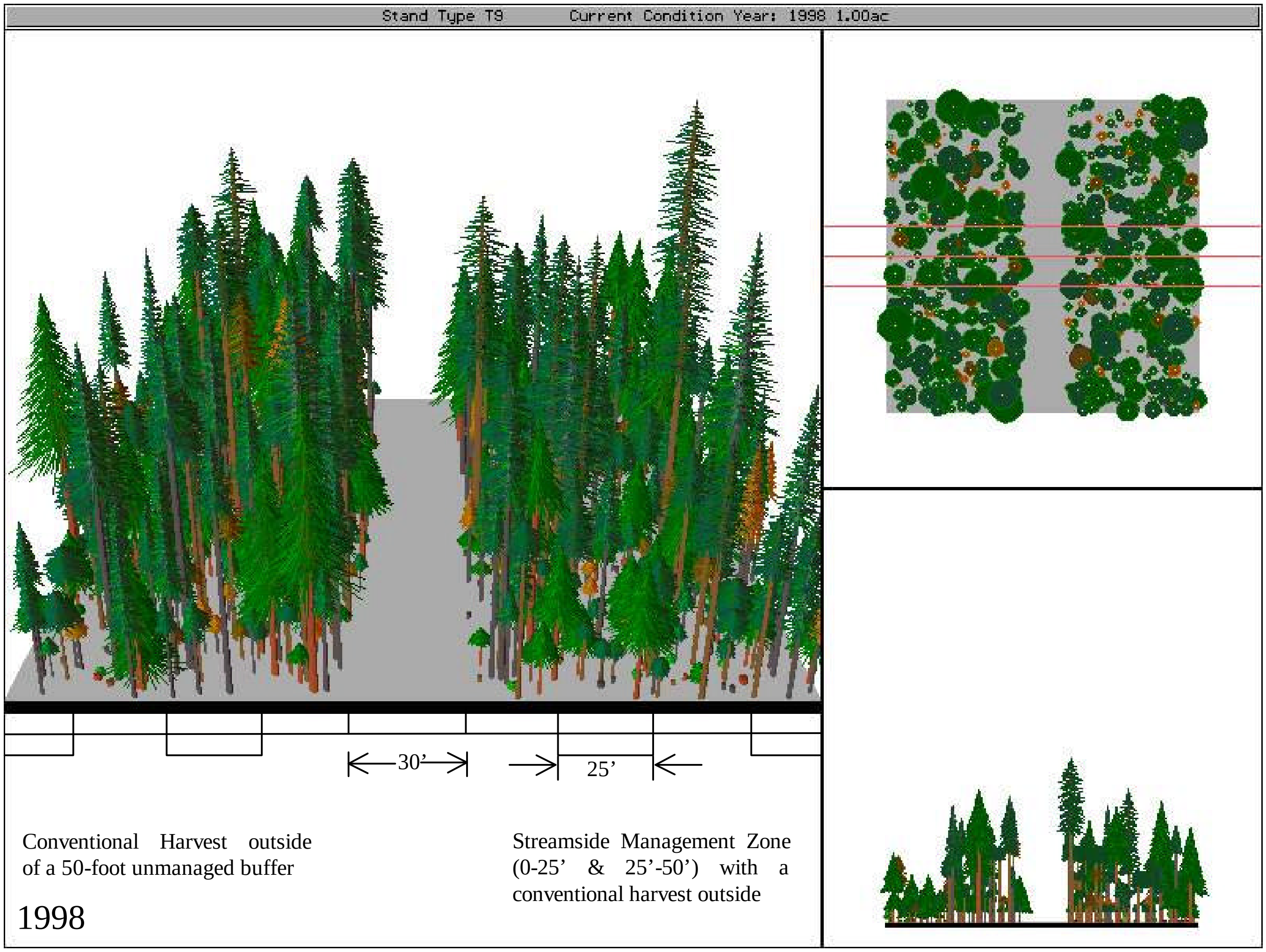
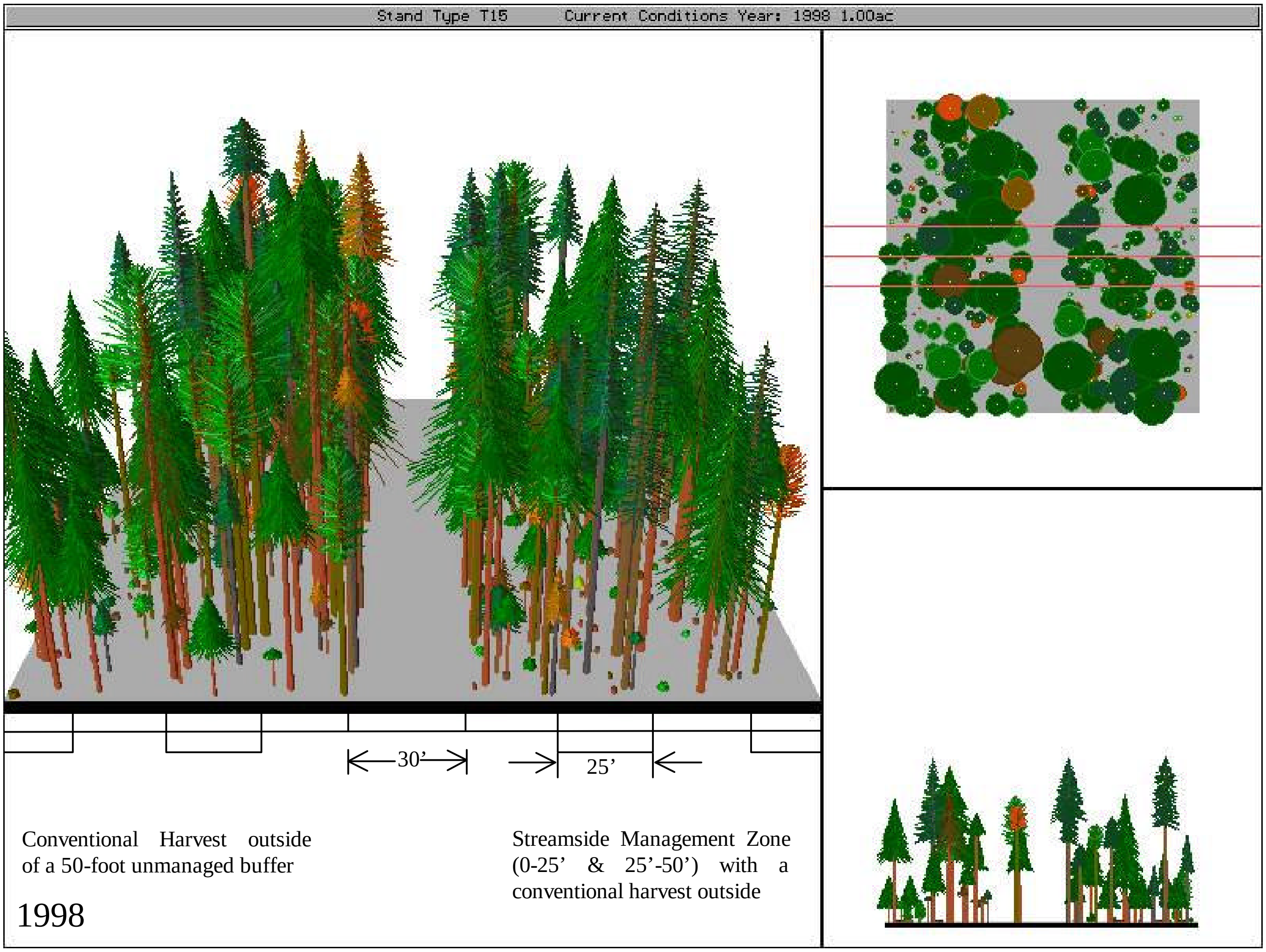


Figure 28. Visual representation of riparian stand type T15, showing the character of the riparian forest after harvest according to Montana's SMZ guidelines for fish-bearing (Class 1) streams, and the character of the adjacent forest after conventional single tree selection harvest.



4.5 A Review of Functions

The maintenance and recovery of fish habitat in the project area is not solely governed by LWD. As mentioned earlier, riparian areas provide other ecological functions, including bank integrity, stream temperature moderation via canopy closure, nutrients, sediment filtration and flood-flow energy dissipation (Harmon et al. 1986, Meehan 1991). In the introduction we proposed that of all the key riparian functions, LWD recruitment would require the widest riparian management areas to achieve desired outcomes. The LWD recruitment evaluations showed that a riparian management zone with a width approximately equal to half a site-potential tree height is capable of providing sustained LWD inputs and maintaining in-channel LWD loads near natural levels. Furthermore, these outcomes can be achieved with varying degrees of timber harvest within the riparian area. Are other functions besides LWD recruitment protected within this distance, even with some amount of timber harvest? We believe that there are situations where LWD is not the most limiting factor with respect to widths of riparian management zones. These situations are described in the following review of other key riparian functions.

4.5.1 Bank Integrity

Bank integrity contributes to Clean and Complex Habitat, and indirectly to Cold Habitat. Bank erosion is a natural and important habitat-forming process. As was shown, bank erosion can be a significant source of LWD (see also Benda and Sias 1998), and is probably the primary source of very large LWD pieces with attached rootwads. In larger, lower gradient streams, undercut banks in forest and meadow riparian areas can provide high quality salmonid rearing habitat as well as cover for adult salmon and bull trout en route to and during spawning. Excessive bank erosion could lead to impairment of the "cool" function if stream widening and shallowing from sediment accumulation results in stream heating. Also, trees and other vegetation that provide bank protection are likely to be important contributors to canopy closure and litterfall.

Protection of the bank integrity function requires maintenance of streambank vegetation. If enough trees or other vegetation are maintained after timber harvest or livestock grazing to provide and sustain an interlocking root network, then bank integrity should be protected. The strip of vegetation should be wide enough to anticipate channel movement (e.g., the channel migration zone). Note that trees growing in or near the streambank also have a relatively high probability of entering the channel and becoming LWD. Of all the key riparian functions, bank integrity requires the least distance to achieve 100% function.

4.5.2 Litter Fall and Nutrient Inputs

Leaf litter and other organic detritus from riparian vegetation is not directly related to the “four Cs” but is important as an energy source for the aquatic food web. In addition to surface inputs, dissolved nutrients may reach the stream through shallow groundwater or hyporheic flow paths. Little quantitative information is available to describe the relationship between cumulative inputs of these materials and distance from the stream. Culp and Davies (1983) found that compared with complete clearcutting, a 10 m (33 ft) riparian buffer would provide leaf litter inputs to streams that were similar to pre-logging conditions. We surmise that, like bank integrity and canopy closure, the greatest amount of inputs are derived from vegetation rooted immediately adjacent to, or very near the stream banks, and possibly greater distances in channel migration zones with pronounced hyporheic flow. Riparian management strategies that focus on LWD recruitment should provide the bulk of terrestrial nutrients to the stream system. Note that different tree species vary in their importance for nutrient production, and when present in riparian areas, a mix of deciduous and conifer species should be maintained.

4.5.3 Canopy Closure and Stream Temperature

Riparian vegetation that provides canopy closure over the stream has a well-established role in keeping streams Cold (Beschta et al. 1987). Because direct-beam solar radiation is a principal source of heat energy inputs to streams (Brown 1969), it follows that vegetation immediately adjacent to streams provides the greatest relative benefits for intercepting solar radiation. Vegetation farther from the stream may increase the density of the canopy and thereby help reduce solar inputs, but these benefits diminish with increasing distance from the channel. With this in mind, an appropriate management question is: What practices lead to little or no measurable decreases in canopy closure after timber harvest in riparian areas? For this answer we turn to a combination of field data and model results. Our best information regarding the effects of timber harvest on canopy closure in Montana and Idaho comes from Plum Creek field studies. In pre- and post-harvest surveys of the same (Montana; Sugden et al. 1998) or similar (Idaho; Greg Watson, unpublished data) stream reaches, statistically significantly decreases were detected in four of twenty reaches. It is not known whether the measured canopy reductions (7%-13%) were enough to cause streams to warm to stressful levels for aquatic biota.

In western Washington, no pre- and post-harvest data were available, so instead we used the canopy closure estimation feature of the RAIS model (Welty 1998). Modeled canopy closure ranged between 50% and 70% on a 10-ft wide channel for all options (including no harvest to 175 ft). Option 2 (25-ft no harvest and 50-ft managed) provided the least amount of canopy closure, but only 10% less than the 175-ft no-harvest scenario. Options 3 and 4 with a 50-ft no-harvest core area provided intermediate amounts of canopy closure (within 5% of unmanaged maximums). We conclude from this that in the majority of cases, existing riparian protection guidelines in Montana and Idaho, or a 50-ft no-harvest buffer in western Washington are sufficient to result in little appreciable change in canopy closure on streams after harvest. The Montana results suggest that in some cases, current harvest guidelines within 50-ft of the stream can reduce canopy closure levels. These appeared to be instances with the greatest amount of harvest (>30%) in the SMZ (Sugden et al. 1998). Additional effectiveness monitoring work is needed to better quantify how often this situation occurs, and its effect on stream temperatures.

Having established that canopy closure can be minimally affected within 50-ft of a stream, the next question is whether vegetation beyond this width can affect stream temperatures via changes in microclimate. Trees and other vegetation within riparian areas can influence local microclimate. Air temperature, relative humidity, wind speed, solar radiation, soil moisture, and surface soil temperatures have all been shown to respond to complete removal of the trees adjacent to upland (Chen et al. 1995) and riparian (Brosofske et al. 1997) forests. In a study of microclimate changes near streams with buffer widths that ranged from 17 - 72 m (56 -236 ft), Brosofske et al. (1997) found no relationship between stream buffer width and stream temperature, except in one case where streamside trees were nearly absent. This corroborates studies of the effects of clearcutting on stream temperatures (e.g., Brown and Krygier 1970), and supports our earlier conclusion about the importance of canopy closure from vegetation immediately adjacent to the channel. However, Brosofske et al. (1997) also reported a strong correlation between near-surface (~2 inches below mineral soil) soil temperature and stream temperature at one station located 197 ft (60m) from the edge of the riparian buffer in a recent clearcut. Though they did not find the same result at other stations along the same transect, or at other transects, there may be situations where canopy removal via timber harvest could result in warming of very shallow groundwater. Until further research identifies the nature of these situations and establishes a cause-effect relationship between upland timber harvest and stream temperatures, managers may want to exercise caution when operating equipment around wetlands and obvious areas of near-surface groundwater. Partial harvest techniques should ameliorate this issue by retaining forest canopy after harvest.

In our review of the LWD functions in non fish-bearing streams of western Washington, we suggested that openings in riparian areas for yarding corridors would not compromise LWD functions. Does this conclusion hold true for maintaining stream temperature? Caldwell et al. (1991) identified field situations where larger tributaries (those that contribute more than 20% of the flow of the receiving stream), if left without canopy closure, led to warming of water in the receiving streams. They suggested that this effect could be diminished provided the tributaries flowed through approximately 500 ft of stream with canopy closure before reaching the stream of concern. More recently, Hetrick et al. (1998) and Keith et al. (1998) demonstrated a more rapid response of stream temperatures to changes in canopy closure. In their study of southeast Alaska streams, they found increases and decreases of several degrees (°C) as streams flowed through alternating 130-230 ft lengths of open and closed canopies, respectively. All of these findings comport with principles of stream heating and cooling outlined by Adams and Sullivan (1990). We conclude that potential adverse warming of fish-bearing streams from perennial non fish-bearing can be prevented by providing a “thermal buffer” for approximately 500 ft. upstream of the confluence with fish-bearing streams. This would apply to tributaries that contributes more than 20% of the flow to the fish-bearing stream. At present, this would be necessary only in Washington and some stream reaches in Idaho.

4.5.4 Sediment Interception

Fine sediment delivered to streams from and through riparian areas primarily affects the Clean aspect of fish habitat. Two sources of sediment involve riparian areas: discrete sources from mass wasting and diffuse sources from disturbed soil in and near riparian areas. Sediment inputs from mass wasting should be addressed by other processes that identify slope stability hazards and provide prescriptive measures (e.g., via Watershed Analysis and site-specific actions). Sediment inputs from diffuse sources such as erosion from compacted or disturbed soil due to forest roads or heavy equipment can be delivered to streams through sheetwash and rilling. Riparian areas can serve as an important interception zone for diffuse sources of sediment. Vegetation and downed woody debris act as obstructions that reduce sediment travel distance (Brake et al. 1997; Ketcheson and Megahan 1996; Burroughs and King 1989, Packer 1967).

In the Oregon Coast Range, Brake et al. (1997) found mean travel distances of 16.7 and 30.6 feet from culverts draining old and new roads, respectively. In highly erodible granitic soils of Idaho, Megahan and Ketcheson (1996) found mean sediment travel distances of 173, 39 and 21 feet for culverts, rock drains and fillslopes, respectively. Also in Idaho, Burroughs and King (1989) found average sediment travel distances ranging from 26 to 80 feet for fillslopes with varying amounts of flow contribution from culverts. Adding logging slash barriers to the fillslopes significantly reduced average sediment travel distance to 4 feet. Packer (1967) found that hillslope obstructions explained one-third of the variability in sediment travel distances below roads. Obstructions on the ground clearly play a large role in sediment interception. Heatherington (1976) also found that a buffer strip prevented sediment transport to streams, whereas nearby streams without buffers had sediment delivered where soils were disturbed. In Montana, Pfister and Sherwood (1991) determined that a streamside management zone of 50 ft would suffice for sediment interception except where wetlands, steep slopes, or erodible soils occurred. The most effective strategy for reducing management-related inputs of sediment to streams is to stop erosion at the source (McGreer et al. 1998). However, for any sediment that is generated in and near riparian areas, these studies suggest that the distances needed to achieve LWD functions will also in most cases minimize or eliminate delivery of the sediment to streams.

4.5.5 Riparian Management Strategies for Maintaining Ecological Functions

The riparian width needed to meet the full suite of ecological functions is largely dictated by large woody debris requirements because of the constraints imposed by the relatively large amount of time and space necessary to create enough trees of a functional size. As shown above, other important ecological functions such as bank integrity and leaf litter inputs typically will be provided by buffers targeting LWD and shade. Our work indicates that in most instances, a riparian management area within fifty feet of streams in the interior Columbia River Basin can provide functions necessary to maintain fish habitat and healthy fish populations. Additional protection is provided by the single-tree selection practices commonly used beyond 50 ft. Trees and other vegetation outside the SMZ will provide additional litter inputs, flood flow energy dissipation, windthrow protection, and riparian microclimate.

Wider buffers may be needed for specific channel types that are particularly responsive to loss of riparian trees and for western Washington streams because of the potential for larger trees west of the Cascade Crest and more intense runoff that creates greater interaction between stream channels and LWD.

Existing rules in Montana, Idaho, and Washington allow the harvest of trees within 25 ft of streams. These trees have a relatively high probability of being recruited to the channel, and therefore careful consideration should be given to channels where LWD is a critical determinant of the amount and quality of fish habitat. For example, plane-bed /forced pool-riffle channel types (1.5-3% gradient) may require more protection than existing State Rules to maintain good quality habitat. Also, channel migration zones, particularly types B and C (Appendix A), are not likely to be adequately protected with standard 50-ft riparian management zones. These sensitive channel reaches were identified as warranting the highest level of protection so that more trees are retained closer to the stream for bank stability and shade. Also, failure to recognize a CMZ may mean that the stream migrates outside the normal riparian management zone, which could defeat the purpose of riparian protections.

Non-fish streams in Washington and Idaho currently have limited protection (Sugden and Light 1998). This can be corrected with a streamside leave tree strategy that focuses on the functions provided by these channels. Because streams can warm if the canopy is removed via timber harvest, a "thermal buffering strategy" could be considered to ensure tributary water temperatures are kept within acceptable limits before reaching fish-bearing waters.

Desired levels of canopy closure may not be achieved under existing regulations in Idaho and Washington. As mentioned earlier, Idaho's existing rules are expected to generate lower amounts of LWD. Idaho's stream shading requirements will provide an indeterminate level of protection for Class 1 streams. The adequacy of this level of shade retention to meet watertemperature standards in Idaho is not known. As discussed in Sugden et al. (1998), many non fish-bearing (Class2) streams in the project area in Idaho have been designated as "streams segments of concern," which increases the shade retention requirements to Class 1 stream levels. Heavy equipment restrictions and canopy closure on stream-connected wetlands or very shallow groundwater with potential to influence stream temperature should also be considered.

Bank integrity may need special attention in areas with excessive livestock grazing (particularly in treeless [meadowed] reaches, and in unbuffered reaches of perennial and seasonal headwater streams in Washington and Idaho.

4.6 Monitoring and Adaptive Management

As mentioned earlier, despite uncertainties regarding some aspects of riparian structure, function, and response to management, we are reasonably confident that available scientific information enables managers to make appropriate decisions regarding economic and ecologic balance in riparian areas. While management strategies derived from this information for the HCP are being implemented, remaining uncertainties can be addressed through research and monitoring. To this end, the Riparian Cause-Effect Pathway is a helpful tool. By partitioning out the elements that lie along the pathway, effectiveness monitoring can be more fruitful and cost-effective. When a practice is determined to be ineffective, then the pathway enables the investigators to establish why. Was the recommended practice complied with? Was the technical basis for the recommendation wrong? If the practice was appropriate, and it was implemented as designed, was there some other factor that contributed to failure? These and other pertinent effectiveness questions can be answered for both site-level practices and watershed-level management systems (Schuett-Hames et al. 1998).

Another important element of monitoring for adaptive management is situational context. Situational context is the information needed to address natural variability and to “separate the signal from the noise.” Also, the numerous assumptions used in the modeling efforts warrant validation and refinement. Identified patterns of stream channel sensitivities, distribution of fish species and their life history stages, characteristics and trajectories of dominant riparian stand types, and stand-specific in-channel LWD targets, all offer opportunities for testing and refinement through adaptive management. These topics are discussed further by Hicks et al. (1999).

4.7 Opportunities for Stream Ecosystem Improvements

Of the nine dominant stand types, five offered little to no near-term harvest opportunities (Appendix C), and none generated substantial LWD loads for decades even when left unmanaged (Table 11). For those stands where past harvest practices contributed to the current condition, silvicultural treatments may be helpful to accelerate regrowth of riparian trees. Where LWD has been depleted, a situation more commonly encountered in western Washington than elsewhere in the project area (e.g., see Huntington 1995, Richmond and Fausch 1995) silvicultural treatments (Berg 1995), or in-channel LWD placement, may aid rapid re-establishment of in-channel conditions. It is infeasible to recommend general actions for this purpose, owing to the site-specific nature of situations where restoration opportunities exist. However, the spectrum of stream channel sensitivities displayed in Figure 6 serves equally well to guide prioritization of restoration efforts; plane-bed channels that can be converted to forced pool-riffle channels by addition of LWD are prime candidates for this type of restoration effort.

Whenever deliberate manipulation of channel features is contemplated as a means of improving fish habitat, however, appropriate experts should be consulted. Past attempts to “improve” fish habitat have in some cases produced the opposite result (e.g., Hall and Baker 1991, Frissell and Nawa 1992). Also, when considering silvicultural manipulation of stands, efforts to improve one function (e.g., LWD), may negatively affect others (e.g., canopy closure).

5.0 CONCLUSIONS

This report provides forest managers with the information needed to develop riparian management strategies that maintain and enhance fish habitat and the stream ecosystem that supports native fish populations. Slightly different strategies were recommended for the project area east and west of the Cascade Mountain Crest, owing to differences in watershed characteristics and timber harvest methods (e.g., partial vs. clearcut harvest). Key findings are presented below:

5.1 Approach and Conceptual Framework

- Tailoring management actions to address specific riparian situations was deemed the best way to achieve balance between economic and ecologic objectives.
- Large woody debris was identified as an important element of the structure and function of stream ecosystems. LWD plays a significant role throughout the drainage network.
- Channel gradient classes (Montgomery and Buffington 1997) were used as a practical means of assigning protection priority to all stream reaches in the project area. Sensitivity to LWD provided the basis for prioritization.
- Stream channels with gradients between 1.5 and 3% were identified as the most sensitive to LWD gain or loss, and were therefore assigned the highest priority for protection.
- Channel migration zones were identified as additional areas with a high sensitivity to LWD (O'Connor and Watson 1998).
- The recommended levels of riparian protection are highest in lower gradient where concentrations of native salmonids are greatest.
- Non fish-bearing tributaries help moderate the rates and magnitudes of stream temperatures, sediment levels, and nutrients.

5.2 Evaluating Riparian Management Strategies

- LWD loads in unmanaged streams of the project area were used to set targets in fish-bearing and non fish-bearing streams.
  - For interior Columbia River Basin streams, a mean of 78 LWD pieces (±57 S.D.) per 1,000 ft of stream length was selected as a target.
  - For western Washington streams, a mean of 98 LWD pieces (±62 S.D) per 1,000 ft of stream length was selected as a target.
  - For smaller and steeper streams without fish, the target LWD load was 163pieces per 1,000 ft of stream length. This applies throughout the project area.

- Riparian stand characteristics (tree size, density, & species) suggest nine dominant stand types occur within Tier 1 basins of the project area.
  - five of the nine stands are not immediately harvestable under montana SMZ guidelines.
  - four of the nine stands offer immediate or near-term harvest opportunities.
  - Montana SMZ laws will provide streams with adequate LWD for two of the four harvestable stands.
  - The remaining two harvestable stands (H9 and H15) may require additional leave trees within 50 ft to attain the LWD target.
  - Existing regulations in Idaho and Washington may not be sufficient to achieve LWD and shade targets. This is especially true for perennial, non fish-bearing streams in Washington.
- With respect to the width of riparian management area needed to provide adequate levels of key riparian functions (LWD recruitment, canopy closure/microclimate, streambank integrity, and nutrient inputs), LWD recruitment was the most limiting. For fish-bearing streams in the interior Columbia River Basin, approximately 85% of LWD inputs (minimum qualifying piece dimensions of 10 cm x 2 m) were estimated to be derived from within fifty feet of the channel. In western Washington, better growing conditions increase this distance to approximately 85 ft. Inputs of most larger LWD, including "key" pieces, was estimated to occur within much shorter distances.
- Functions provided to fish-bearing streams by non fish-bearing streams can be maintained by ensuring (1) LWD continues to enter these channels where it can trap sediment, reduce flow velocities, and contribute to the nutrient cycling, (2) sufficient canopy/closure shading is provided to cool streams to acceptable temperatures by the time their water reaches fish-bearing reaches.
  - In non fish-bearing streams of the interior Columbia River Basin, Montana's existing Best Management Practices, and Idaho's Streamside Protection Zone laws were judged to adequately protect riparian functions along this class of stream. Existing rules in Washington were deemed inadequate to provide these functions.
- The functions provided by continuous buffers along non fish-bearing streams in the interior Columbia River Basin where partial harvest methods are used can be achieved in even-aged management systems in western Washington using a continuous buffer strategy with allowances for yarding corridors.
- Estimated trends in LWD loads are positive for all riparian stand types and stream classes managed using Montana's SMZ guidelines.
- There may be opportunities to improve fish habitat conditions or stream ecosystem processes through silvicultural practices or in-channel LWD placement.
- Monitoring can be used to validate assumptions used in modeling efforts and to measure effectiveness of riparian management practices. The monitoring results can be used for refining the practices via adaptive management.

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APPENDIX A

GEOMORPHOLOGY OF CHANNEL MIGRATION ZONES AND IMPLICATIONS FOR RIPARIAN FOREST MANAGEMENT

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TABLE OF CONTENTS

1.0 INTRODUCTION

The concept of channel migration zones (CMZs) is a consequence of the fact that stream channels are dynamic features of the landscape that change position on valley floors under some circumstances. CMZs may be defined as terraces and/or floodplain areas adjacent to stream channels that have a high likelihood of being occupied by the stream channel at some time in the foreseeable future. The Washington Forest Practices Board (1995) defined the channel migration zone as "...the area that streams have recently occupied (in the last few years or less often decades), and would reasonably be expected to occupy again in the near future." Lateral shifts of stream channels may occur suddenly as the result of flood flows and/or reduced channel capacity caused by sedimentation or debris jams, or gradually as a stream erodes the outside edge of meander bends.

Identification of CMZs and adoption of specialized management practices in these areas will reduce potential long-term adverse effects of riparian forestry on aquatic ecosystems. Existing regulations relating to forestry activities in riparian zones typically establish limits on harvest activities near stream channels based on the location of ordinary high water (OHW). These regulations are generally intended to provide for shading of the stream surface and recruitment of large woody debris (LWD) to stream channels. CMZs require specialized management because existing regulations may fail, in some areas, to provide the desired levels of shade or LWD recruitment following episodes of channel migration.

2.0 CHANNEL MIGRATION PROCESSES

Lateral shifts in stream channels occur both suddenly and gradually. When a stream channel *avulses*, that is, when streamflow spills out of the banks of an existing channels, a new channel may be eroded in a short period of time. Channel avulsion typically occurs when the existing channel is incapable of carrying all of the water and sediment supplied to it.

During periods of high stream flow or floods, avulsions may occur at locations where channel morphology forces flow velocity to decrease and flow depth to increase. This often occurs at sharp bends in the channel. If the flow depth exceeds the height of the bank on the outside of the bend, water may spill over the bank. This water may then spread over a floodplain surface or, if the valley has a relatively steep slope and topographic features that concentrate the overbank flow, it may erode a new channel.

Channel avulsions are also caused by deposition of sediment and/or LWD that reduce channel capacity and increase flow depth (Keller and Swanson, 1979). Formation of large gravel bars or debris jams may also re-direct high velocity streamflow toward banks. When such flows encounter erodible banks and/or relatively low-lying terrace surfaces, a new channel may form. In some cases, the new channel will accommodate all of the streamflow and the former channel may be abandoned. In other cases, the new channel may carry excess flows only and the existing channel maintains its character. In still other cases, both new and existing channels may carry water under base flow conditions.

Gradual changes in channel position also occur, most often in low-gradient channels (approximately < 1% slope), as the outer bank of a meander bend is eroded during periods of routine (e.g. annual) peak flow. Bank materials for these low-gradient channels tend to be fine-grained, and are therefore susceptible to erosion. Avulsions may also occur in these types of streams, however, terrace surfaces adjoining such low-gradient streams tend to be nearly level and overbank flow is more likely to spread on a floodplain where there is insufficient slope to erode a new channel.

Gradual channel migration as conceived here occurs during most years at annual rates that are a fraction of the bankfull channel width. In contrast, channel migration by avulsion is conceived to occur sporadically and result in lateral shifts in channel position ranging in size from the bankfull channel width to the valley width. With respect to the former process, the rate of channel migration may be slow enough to allow harvest and regeneration of forest stands in riparian areas subject to channel migration. In the case of the latter process, channel migration is sudden and can occur anywhere within the CMZ. Consequently, if maintenance of aquatic ecosystem structure associated with channels in CMZs is a priority, riparian forest conditions must be maintained in a state that would allow new channels to recruit significant quantities of LWD.

3.0 OCCURRENCE OF CMZS AS A FUNCTION OF CHANNEL TYPE

In order for channel migration to occur, the valley width must be sufficiently large to accommodate lateral shifts in position. Channels that are confined by valley walls are not prone to channel migration. The definition of a confined channel in common usage relates the bankfull channel width to the valley width. Rosgen (1994) defined the “flood prone width” as the width of a surface perpendicular to the trend of the valley or channel defined at an elevation twice that of the bankfull depth (corresponding to a flow with recurrence interval of about 1.5 to 2 years). The flood prone width corresponds to elevations of relatively frequent floods (< 50 yr recurrence interval). The ratio of flood prone width to bankfull width is the entrenchment ratio.

Entrenched channels have an entrenchment ratio (ER)$\leq$ 1.4, moderately entrenched channels have 1.4 < ER $\leq$ 2.2, and slightly entrenched channels have ER > 2.2 (Rosgen, 1994). Interpreted for purposes of predicting potential channel migration, slightly entrenched channels (Rosgen Types E, C, D and DA) have the greatest potential for channel migration, while moderately entrenched channels (Rosgen Type B) have modest potential. Entrenched channels (Rosgen Types A, F and G) have little or no potential for channel migration.

Using Rosgen's system of channel classification, channels prone to channel migration might include Types E, C, D and DA, which are defined to have slopes < 2% (Rosgen 1994). Type B channels with some channel migration potential range in slope from 2% to 4%. Hence, channels with slope > 4% could be excluded on the basis of interpretation of Rosgen's classification system.

An alternative to Rosgen is the channel classification system defined for watershed analysis by the State of Washington (Washington Forest Practices Board, 1995). In this system, confined channels are defined to be those in which the ratio of valley floor or floodplain width to channel width is < 2. Channels with moderate confinement have ratios between 2 and 4, and unconfined channels have values ratios > 4. The means to define the valley width in this system are not precisely defined, leading to a somewhat ambiguous classification. However, using this system, channels with valley width to channel width ratios > 2 could be expected to have potential for channel migration.

The Washington method classifies stream channels according to slope classes that roughly correspond to channel morphologic types defined by Montgomery and Buffington (1993). These slope classes are < 1%, 1-2%, 2-4%, 4-8%, 8-20% and > 20%. Subsequent research by Montgomery and Buffington (1997) redefined the slope classes corresponding to channel geomorphologic types as <1.5% (pool-riffle), 1.5-3% (plane-bed), 3-6.5% (step-pool), and >6.5% (cascade). Field observations by the author suggest that channel migration occurs in valleys as steep as the 8-20% slope class, typically in reaches where channel slope declines relative to areas upstream. However, as slope increases, valley confinement and channel entrenchment generally increase and thereby limit the horizontal range of migration. In other words, valley width is rarely >2 bankfull width in channels with slopes >8%. Field observations by the authors suggest that significant channel migration occurs infrequently where channel slope > 6%. On the basis of field experience and the Washington DNR watershed analysis classification system, CMZs would be limited to channels with slopes < 8% with moderately confined or unconfined boundaries.

For the purposes of investigating hypotheses proposed regarding CMZ processes, it is suggested that the entrenchment ratio definition of Rosgen (1994) be used in conjunction with slightly modified channel morphologic slope classes defined by Montgomery and Buffington (1993, 1997). Rosgen's entrenchment ratio is well-defined, and is a sensitive indicator of potential lateral flow dispersion that is required for channel migration. Montgomery and Buffington's channel classification system provides more complete (compared to Rosgen), description of channels steeper than 4%, and more explicitly integrates fluvial geomorphology and watershed process with channel classification. These slope classes are modified for classification of CMZs (Table 2). Channels steeper than 8% are assumed to be sufficiently entrenched that the channel position cannot shift.

Table A1. Hypothesized likelihood of significant channel migration as a function of channel entrenchment (confinement) and supply of sediment and/or LWD.

Entrenchment Ratio	Low Sediment/LWD Supply	High Sediment/LWD Supply
ER > 2.2 (unconfined)	Moderate	High
1.4 < ER ≤ 2.2 (intermediate confinement)	Low-Moderate	Moderate-High
ER ≤ 1.4 (confined)	Low	Low

4.0 INFLUENCE OF DISTURBANCE REGIMES ON CHANNEL MIGRATION PROCESSES

As noted earlier, channel migration by avulsion, and to some extent gradual migration by bank erosion, occur in response to deposition of coarse sediment or LWD in stream channels that locally reduce channel conveyance, forcing streamflow out of the existing channel. Channel migration in a particular stream reach subject to significant inputs of sediment or LWD will tend to occur if the channel is relatively unconfined (i.e. the ER is relatively large). Thus, in watersheds with higher rates of coarse sediment input, a higher incidence of channel migration might be expected in stream reaches with high ER's (Table A1). Reaches with high rates of LWD recruitment and/or LWD deposition and high ER's would also be expected to have higher incidence of channel migration. Channels with a large ER in watersheds with relatively low input rates for coarse sediment or LWD would be expected to have a significantly lower incidence of channel migration. Finally, channels with small ER's and low rates of LWD recruitment and low supply of coarse sediment would be expected to have no significant channel migration.

The geomorphic development of stream reaches with significant channel migration zones probably occurs over relatively short periods of geologic time (thousands of years). Channel migration processes require a relatively wide valley bottom with an elevation near that of the channel bed (i.e. a high ER). These conditions typically occur when the supply of coarse sediment is in balance with or greater than the stream's transport capacity for coarse sediment. When sediment supply exceeds transport capacity, stream channels aggrade (bed elevation increases), and tend to migrate laterally across the valley floor, depositing sediment on the floodplain. If sediment supply is interrupted, or transport capacity (streamflow) increases, stream channels begin to incise the valley deposits and become entrenched, and former floodplains may become terraces that are infrequently flooded or entirely isolated from the stream. Channel migration diminishes or ceases.

In the case of sediment supply ≥ transport capacity, positive feedback tends to maintain conditions favorable to channel migration. Assuming a high ER, it is likely that a high water table will exist at least seasonally in the floodplain. The high water table increases the likelihood of significant blowdown of trees. Recruitment of entire trees by this mechanism increases the potential for debris jam formation that induces channel avulsion. Channel avulsion also recruits LWD. High LWD concentration in stream channels increases channel roughness and decreases sediment transport capacity. Under these circumstances, it is possible that LWD recruitment to the channel may to some extent compensate for reductions in sediment supply and maintain conditions favorable to channel migration.

Historic changes in sediment supply and regional hydrology in many areas of the northwestern United States are related to the retreat of glaciers in the past 10,000 to 100,000 years. In most areas, stream channels have incised valley floor deposits and created one or more sets of terraces that are no longer subject to inundation. Thus, regional conditions suggest that incidence of channel migration should be diminishing. Areas where high LWD recruitment occurs may therefore be locations where channel migration processes are maintained, despite regional geologic trends that reduce potential for channel migration.

If this hypothesis is correct, decreases in long-term LWD recruitment to certain stream channels could induce essentially irreversible changes in channel morphology. Decreases in LWD recruitment could occur as the result of stand-replacing fires, logging, or other disturbance to riparian forest stands. In terms of gross channel morphology (and fish habitat), existing forced pool-riffle channels in reaches with active channel migration would be gradually transformed to entrenched plane-bed reaches. Changes in significant fish habitat variables likely would include increased particle size of stream bed sediment, reduced frequency and size of gravel bars, a decrease in the frequency and depth of pools, and decreased variation in stream velocity and depth.

Table A2. Classification of channel migration processes by stream channel slope and confinement classes and hypotheses regarding frequency and process.

Confinement or Entrenchment	Low Gradient < 1.5 % Channel Slope	Moderate Gradient 1 - 6 % Channel Slope	High Gradient 4 - 8 % Channel Slope
Confined/ Entrenched (ER ≤ 1.4)	Uncommon channel type; no hypotheses regarding channel migration processes	Channel migration unlikely	Channel migration unlikely
Moderately Confined/ Moderately Entrenched (1.4 < ER ≤ 2.2)	Uncommon channel type; no hypotheses regarding channel migration processes	*CMZ C (3-6%)* Channel migration by avulsion is not uncommon, but is likely to be spatially-discontinuous, depending on local variation in valley slope and width and disturbance regime	*CMZ E* Channel migration by avulsion is uncommon, and is likely related to debris flows and torrents; may be locally significant depending on local variation in valley slope and width and disturbance regime
Unconfined or Slightly Entrenched (ER > 2.2)	*CMZ A (<1.5%)* Channel migration by gradual erosion of meander bends is common; avulsions may also occur. Areas of potential migration are spatially continuous and include much or all of the floodplain.	*CMZ B (1-3.5%)* Channel migration primarily by avulsion, but may include bank erosion. Areas of potential migration are spatially continuous or discontinuous and include much or all of the floodplain.	*CMZ D* Channel migration by avulsion may be common, and is likely to be related to alluvial fan and debris flow fan processes. Process may be spatially discontinuous continuous, or localized, depending on fan-building processes.

Large scale production of coarse sediment by mass wasting, in some cases due to management, has been observed to induce channel aggradation and channel migration (O'Connor and Cundy 1993, O'Connor 1997). This occurs in geologic settings where channels are prone to migration processes, and have been subject to previous cycles of aggradation and degradation. In some areas where recent channel aggradation and migration has been observed, riparian forest stands in the channel migration zone had been converted to younger seral stands by prior timber harvest (O'Connor and Cundy 1993, O'Connor 1997). It is hypothesized that channel and aquatic habitat conditions in these areas would be significantly different had larger trees been present to be recruited when the stream channel began to aggrade. Large diameter, lengthy pieces of LWD with attached rootwads have the highest likelihood of remaining stable in a channel, and can establish stable nodes in the channel. Such “anchors” can promote the development of persistent pools, islands, and side channels that might otherwise be eroded or filled by coarse sediment in a stream reach where an episode of sediment-induced channel migration is occurring.

5.0 IDENTIFICATION AND MAPPING OF CMZS

An hypothetical classification of channel migration potential by channel slope and entrenchment/confinement is presented in Table A2. This preliminary classification is based on prior observations of stream channels in the Pacific Northwest and professional judgement. It is intended as an initial hypothesis to be tested and revised through field observations. Figures 1 though 5 provide summary information and a sketch of representative channel conditions for the five cases in which significant channel migration is hypothesized in Table A2. Summary characteristics of the 5 proposed CMZ types are presented in Table A3 for ease of comparison.

In low-gradient channels (approximately < 1% slope), where channel migration is both gradual and sudden, stream channels are relatively sinuous, stream bends may have a short radius of curvature, and the pattern of channel migration can often be mapped from aerial photographs. Channels of this type are usually wide enough to be visible despite riparian canopy. In many cases, former channel positions may be determined on the basis of topography and/or vegetation, including oxbow lakes. The CMZ in this setting may often be defined in plan form as the area within the meander belt formed by the bends of the river or by the limits of the floodplain. In addition, field identification may benefit from relatively well-defined terraces that correspond to abandoned, inactive terraces within which a lower set of active terraces can be identified that correspond with the CMZ. It may be possible in some cases to quantify channel migration rates from aerial photo records for this channel type. If rates can be calculated, potential management responses can be made more objectively.

In steeper channels (approximately > 1% slope), channel migration is more likely to occur because of sudden channel avulsion. When the channel is wide enough to be visible despite the riparian canopy, it is often possible to observe the position of historic channels. In addition, vegetation types may aid in the identification of channel migration patterns. Mapping of historic channel shifts (where visible in photographs), may serve as the basis for CMZ delineation, however, field surveys may better define the specific circumstances under which channel avulsions occur and thereby narrow the spatial extent of the CMZ.

Stream channels narrower than about 30 to 40 feet typically can be difficult to observe in aerial photographs, depending on riparian forest conditions and the quality and scale of photography. In such areas, evidence of channel migration is difficult to obtain from aerial photographs. Field surveys would generally be required to locate riparian areas where channel migration processes are active.

Table A3. Summary characteristics of hypothesized channel migration zone (CMZ) types.

	CMZ CHARACTERISTIC	TYPE A	TYPE B	TYPE C	TYPE D	TYPE E
Diagnostic Characteristics	Channel Migration Processes	Bank Erosion (primary) and Avulsion (secondary)	Avulsion (primary) and bank erosion (secondary)	Avulsion	Fan-related (alluvial & debris flow fans)	Avulsion (debris flow & debris torrent)
	Entrenchment Ratio	ER > 2.2	ER > 1.4	ER > 1.4	ER > 2.2	1.4 < ER ≤ 2.2
	Slope	< 1.5 %	1 – 3.5 %	3 – 6 %	4 – 8 %	4 – 8 %
	Channel Morphology	Pool-riffle, plane-bed	Forced pool-riffle, plane bed, step-pool	Step-pool, forced pool-riffle plane bed	Step-pool, cascade, plane bed	Step-pool, cascade, plane bed
Supplementary Diagnostic Characteristics	Longitudinal Distribution of CMZ	Continuous	Continuous or Discontinuous	Discontinuous	Continuous or Discontinuous	Discontinuous
	Sinuosity	High	Moderate	Moderate	Low	Low
	Channel Bed Substrate	Silt, Sand, Gravel, Cobble	Gravel, Cobble	Gravel, Cobble, Boulder	Cobble, Boulder	Cobble, Boulder
	Bank Material	Same as bed	Same as bed or coarser	Same as bed or coarser	Typically same as bed, possibly coarser	Typically coarser than bed, possibly the same
	Relative Landscape Frequency	Common	Common	Uncommon	Rare (active fans)	Rare (depends on landslide type & frequency)
Management Sensitivity	Channel Migration Potential (1)	High	High	High	Moderate	Low-Moderate
	CMZ Sensitivity to LWD (2)	Moderate	High	High	Moderate	Moderate
	CMZ Sensitivity to Coarse Sediment	High	High	High	High	Moderate

Notes

1. Channel migration potential is evaluated relative to other CMZ types. Occurrence of channel migration is thought to be a function of both local and upstream disturbance (e.g. Table 1), and variation in local channel slope and confinement within a reach of the given type.

2. The influence of LWD on channel migration processes is affected by channel width. As bankfull channel width increases, a smaller proportion of LWD is likely to have an effect on channel morphology because LWD pieces that are shorter than the bankfull width will tend to be transported downstream. In Washington and Oregon, LWD abundance in stream channels with bankfull widths between 30 and 50 feet decreases significantly compared to narrower channels (e.g. Bilby and Ward, 1991).

Characterization and mapping of areas of active channel migration is recommended in order to distinguish among areas of frequent and infrequent migration, which would likely have significant management implications.

Mapping the spatial and temporal frequency of channel avulsions could provide an objective basis for delineation of CMZs and development of management practices to maintain long-term riparian function. Frequency or rate of channel migration can possibly be accomplished through a combination of aerial photo mapping and field dendrochronology. Quantification of rates or frequency is desirable to provide more objective data for consideration of management responses.

6.0 MANAGEMENT IMPLICATIONS

The preceding sections described CMZ processes and identification. This section discusses potential response to CMZ delineation in the context of riparian forest management. Riparian forests generally have significant existing regulatory protection (e.g. Montana Streamside Management Zone Law). The central objective of the following discussion is to consider whether existing riparian regulations are sufficient for CMZs, and if not, what actions should be considered to enhance the protection of aquatic ecosystem function in CMZs. In general, when existing riparian management zones have a width on each stream bank less than the bankfull channel width, it is much more likely that existing leave tree requirements will be inadequate. If the width of the existing riparian management zone on each bank is a multiple (e.g. 3 or 4) of the bankfull channel width, it is much more likely that existing leave tree requirements will be adequate.

6.1 Case 1: CMZ Width £ Riparian/Streamside Management Zone

An identified CMZ may fall entirely within the limits of existing regulatory streamside or riparian management zones. In these cases, the primary concern is whether existing regulatory requirements for leave trees are sufficient to maintain aquatic ecosystem function in the CMZ. The number, size and location of leave trees should be considered in relation to the number and size of LWD pieces functioning in the CMZ. The management goal should be to ensure that if a channel shift occurs, the number, size and species of LWD pieces recruited to the new channel is comparable to that in existing channels in adequately-functioning comparable channels in comparable CMZs. Potential differences in the importance of LWD in different types of CMZs (see Table 3), might also serve as a criteria. In addition, estimates of frequency and spatial distribution of channel migration events (avulsions) or rate of bank erosion (gradual migration) could be relevant factors in determining appropriate criteria for leave trees.

6.2 Case 2: CMZ Width > Riparian/Streamside Management Zone

An identified CMZ may extend beyond the limits of existing regulatory streamside or riparian management zones. In these cases, the main concerns are the risk of channel migration beyond the existing leave tree zone and the adequacy of existing leave tree requirements for maintenance of aquatic ecosystem function in the CMZ. At a minimum, extending the regulatory requirements for leave trees in the riparian management zone to include the CMZ should be considered. Estimates of the frequency and spatial distribution of channel migration events (avulsions) or rate of bank erosion (gradual migration) that could lead to channel migration beyond the boundary of the existing riparian management zone could be relevant in determining appropriate criteria for leave trees in the CMZ outside the riparian management zone. The number, size and location of leave trees should be considered in relation to the number and size of LWD pieces functioning in the CMZ. The management goal should be to ensure that if a channel shift occurs, the number, size and species of LWD pieces recruited to the new channel is comparable to that in existing channels in adequately-functioning comparable channels in comparable CMZs. Potential differences in the importance of LWD in different types of CMZs (see Table 3), might also serve as a criteria.

7.0 FURTHER RESEARCH

Further research is needed to test the applicability of the proposed CMZ classification system. This should include a literature review, but emphasis should be on field identification and mapping. An investigation of the relationship between LWD abundance, function and size in relation to stream size and CMZ type would help provide criteria for leave trees. Data on the frequency and rate of channel migration by both avulsion and bank erosion would provide relevant perspective on the design of management strategies for maintaining aquatic ecosystem function in CMZs.

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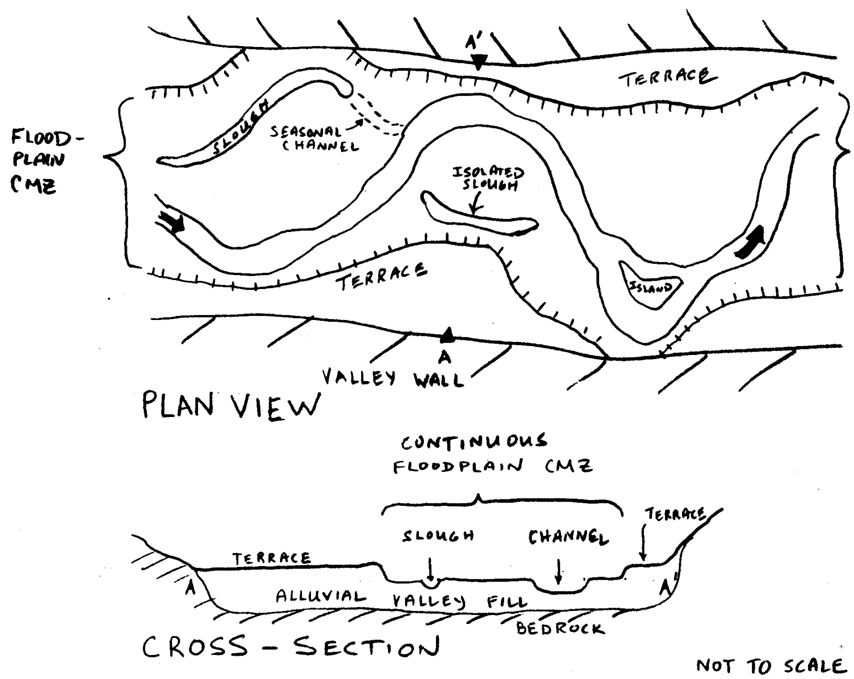
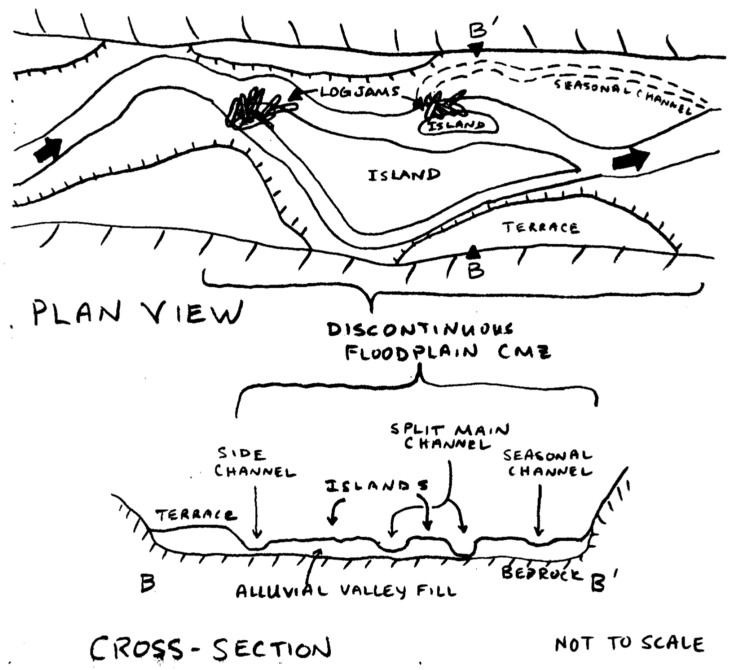
CMZ TYPE A: SLIGHTLY ENTRENCHED, CHANNEL SLOPE < 1.5%



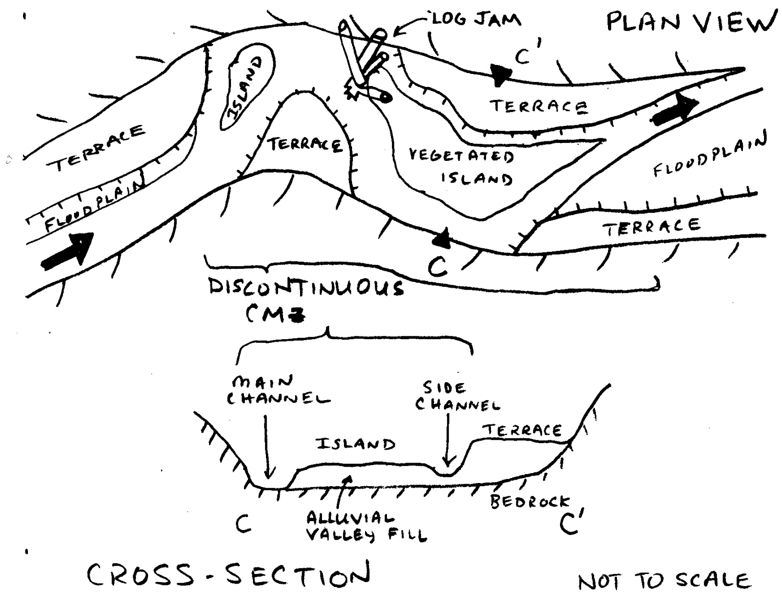
Entrenchment Ratio	> 2.2	**Channel Migration Potential**	High
Slope	< 1.5 %	**Channel Migration Processes**	Gradual erosion (primary) and avulsion
Channel Morphology	Pool-riffle, plane-bed	**Distribution of CMZ**	Continuous across floodplain and parallel channel
Bed Substrate	Silt, Sand, Gravel, Cobble	**Sensitivity to LWD**	Moderate
Sinuosity	High	**Sensitivity to Coarse Sediment**	High

CMZ TYPE B: SLIGHTLY ENTRENCHED, CHANNEL SLOPE 1 - 3.5%



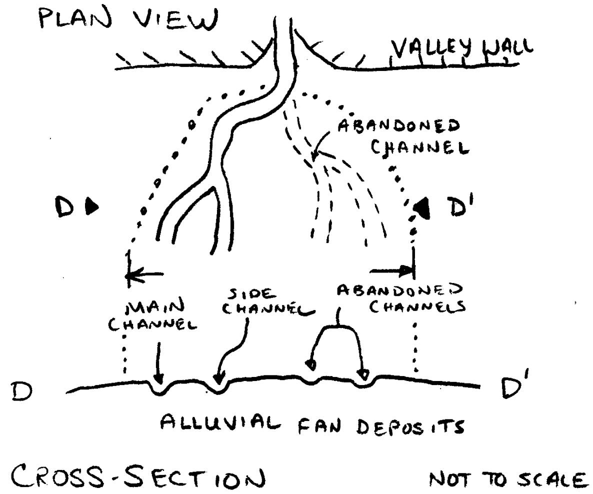
Entrenchment Ratio	> 1.4	**Channel Migration Potential**	High
Slope	1-3.5 %	**Channel Migration Processes**	Avulsion (primary) and bank erosion
Channel Morphology	Forced pool-riffle, plane bed, step pool	**Distribution of CMZ**	Discontinuous or continuous along valley length and width
Bed Substrate	Gravel & Cobble	**Sensitivity to LWD**	High
Sinuosity	Moderate	**Sensitivity to Coarse Sediment**	High

CMZ TYPE C: MODERATELY ENTRENCHED, CHANNEL SLOPE 3 - 6%



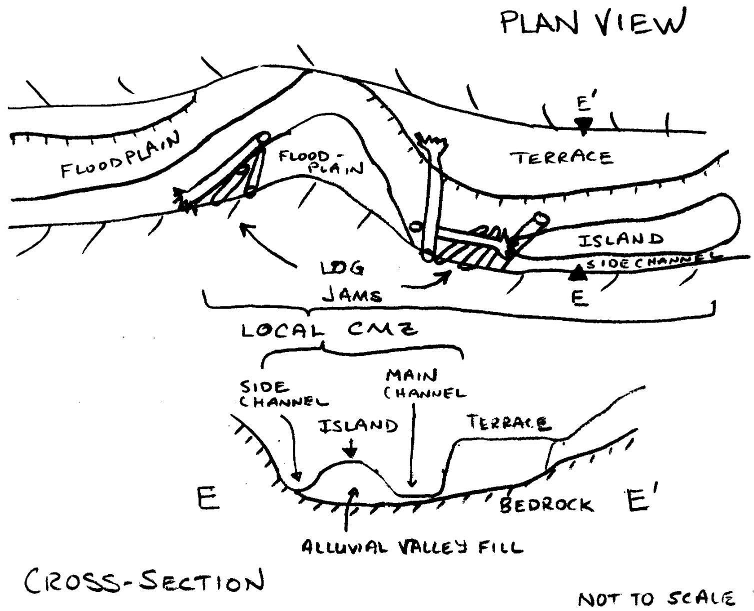
Entrenchment Ratio	ER >1.4	**Channel Migration Potential**	High
Slope	3-6 %	**Channel Migration Processes**	Avulsion
Channel Morphology	Step pool, forced pool-riffle, plane bed	**Distribution of CMZ**	Discontinuous
Bed/Bank Materials	Gravel, Cobble, Boulder	**Sensitivity to LWD**	High
Sinuosity	Moderate	**Sensitivity to Coarse Sediment**	High

CMZ TYPE D: SLIGHTLY ENTRENCHED, CHANNEL SLOPE 4 - 8%



Entrenchment Ratio	> 2.2	**Channel Migration Potential**	Moderate
Slope	4-8 %	**Channel Migration Processes**	Avulsion (alluvial or debris flow fan processes)
Channel Morphology	Step-pool, cascade, plane bed	**Distribution of CMZ**	Continuous or discontinuous
Bed/Bank Materials	Cobble & Boulder	**Sensitivity to LWD**	Moderate
Sinuosity	Low	**Sensitivity to Coarse Sediment**	High

CMZ TYPE E: MODERATELY ENTRENCHED, CHANNEL SLOPE 4 - 8%



Entrenchment Ratio	1.4 < ER ≤ 2.2	**Channel Migration Potential**	Low-Moderate
Slope	4-8 %	**Channel Migration Processes**	Avulsion (debris flow and debris torrent process)
Channel Morphology	Step-pool, cascade, plane bed	**Distribution of CMZ**	Discontinuous
Bed/Bank Materials	Cobble & Boulder	**Sensitivity to LWD**	Moderate
Sinuosity	Low	**Sensitivity to Coarse Sediment**	Moderate

APPENDIX B

TECHNICAL RATIONALE SUPPORTING SELECTION OF IN-CHANNEL LWD TARGETS FOR FISH-BEARING AND NON-FISH-BEARING STREAMS

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TABLE OF CONTENTS

TABLES

FIGURES

1.0 FISH-BEARING STREAMS — INTERIOR NORTHWEST LARGE WOODY DEBRIS (LWD) LOADING RELATIONSHIPS: AN ANALYSIS OF AVAILABLE DATA

1.1 Introduction and Approach

Twelve studies of unmanaged northwest streams and/or databases that included data suitable for this Appendix were obtained for analysis.

Comparison of LWD study results is difficult due to the complex array of factors that affect LWD presence and character, and due to disparity of inventory methods. Differing methodologies include minimum LWD diameter and length inventoried, different criteria for qualifying pieces (total number, or "acting," "functional," or "key" pieces only), pieces in aggregations included as individual pieces or counted only as a single piece, stream size measured as channel width or as wetted width, and miscellaneous other complications. We have identified these metrics for each of the studies included in this review.

To allow rational comparison of the twelve studies, we organized our analysis into a series of questions and systematically examined the data from each of the studies relevant to each question. Statistical differences were examined at the 0.10 level of significance for all relationships tested. For some studies, we did not have the original and complete databases available to us for this review, preventing us from examining some relationships of interest that might be revealed by the complete databases. It may be possible to obtain all or some of these data for examination and inclusion at a later date.

1.2 The Studies

For each of the studies examined, Table B-1 identifies the study author, geographic location of the study, key characteristics of the streams measured, and measurement methods that affect interpretation of the results, including minimum LWD diameter and length criteria.

1.3 Results

Table B-2 lists the questions we posed for each of the interior studies that we examined and provides answer notation for each question/study cell in the table. A "Y" (yes) means that the question is answered affirmatively by the study's data and is statistically significant at the 0.10 level. A "W" (weak) indicates that the relationship is statistically significant, but the correlation (R^2) is very low. To provide a consistent means of making a yes, no, or weak call for the collective body of studies, we assigned a value of 1 to yes, 0.5 to weak, and 0 to no, and calculated the mean of the values for each question. Mean scores of <0.33 indicate that the study's data do not support the conclusion, 0.33 to 0.67 provide weak support, and >0.67 affirm the question.

1.3.1 LWD Frequency

1.3.1.1 What is the frequency of LWD within interior streams in pieces/1,000 feet, and in pieces/channel (or wetted) width?

Table B-3 provides the descriptive statistics for each of the interior studies examined and statistics for westside studies addressed in question 2.

Mean LWD frequency from these studies ranges from a low of 25 (Overton et al., 1995), to a high of 169 (Philbin, 1993). Variability in LWD frequency is high in all of these studies, ranging from approximately one-third to two-thirds of the mean.

Peculiarities of the Overton and Philbin studies explain their low and high mean LWD frequencies, respectively. Overton required LWD to be a minimum of 3 meters in length, or two-thirds of the channel width, whichever was smaller, and reported aggregations of LWD as single pieces. We believe this causes the Overton count to be lower than would have been counted using inventory procedures more commonly used in the rest of the studies examined. Philbin, while using fairly common inventory procedures, focused his study on very small headwater streams, which may have caused LWD frequency to be higher than those found in the other studies. Inclusion of only "acting" LWD in the counts for the Lolo, Huntington, and Bitterroot studies could cause these studies to report lower counts than for studies counting all LWD pieces of qualifying size, although this effect is not apparent for the Lolo. Conversely, inclusion of one-meter LWD should cause the Bitterroot, Knight, and Philbin counts to be high in comparison to studies using 2-meter minimum length criteria, but again, the effect does not appear consistent.

Inconsistent methods caused us to consider adjusting counts to some standard criteria. This was possible only for the Colorado data (Richmond and Fausch, 1995), where 2-meter minimum length criteria could be determined from a histogram of frequency of occurrence of LWD versus length. This histogram indicated that approximately 35% of the LWD measured in this study was between 1 and 2 meters in length. We therefore multiplied the piece count for each stream in the Colorado data set by 0.65 to standardize to 2-meter minimum length criteria before performing our analysis. Data for other studies were not adjusted, due to a lack of any defensible means for deriving adjustment factors.

We performed an analysis of variance (ANOVA) on the piece count data from the five data sets using consistent methodology, i.e., 10 cm diameter and 2 meter length criteria (Sugden, 1998; Watson, 1997, Richmond and Fausch, 1995; Bilby and Wasserman, 1989; Hayes, 1996) and found that the means were significantly different; we were therefore not justified in pooling the piece count results from all five of these studies. However, Duncan's multiple range test revealed that the mean piece counts found for the Swan (Watson, 1997) and Thompson (Sugden, 1998) data sets were not significantly different. Since data for these two watersheds were collected with similar methodology from the same general geographic area, we pooled the Swan and Thompson data for our analysis.

Figure B-1 provides a box and whisker plot comparison of those studies with adequate sample sizes to allow construction of the plots.

1.3.1.2 How does interior LWD frequency compare to westside LWD frequency?

Interior LWD frequency is generally lower than in westside streams; however, due to the aforementioned differences in methodologies, we were unable to test whether these differences are significant (see Tables B-3 and B-4).

1.3.2 LWD / Stream Size Relationships

1.3.2.1 Is interior unmanaged stream LWD diameter smaller than westside diameter?

Yes, LWD diameter is smaller in the interior. LWD diameter data is reported in Table B-5. Interior unmanaged stream LWD diameter is reported in six data sets, ranging from approximately 8 to 11 inches. Diameter for the three westside studies reporting such data ranges from 10 to 21 inches.[29]

1.3.2.2 Is interior unmanaged stream LWD volume (ft^3/1,000 ft) lower than westside volume?

Yes, volume of LWD is lower in interior streams. Volume data is reported in Table B-6. Mean volume for the six interior streams reporting LWD size data ranges from 141 to 2,340 ft^3/1,000 ft. Mean volumes for the two westside studies reporting such data are 4,650 and 6,240 ft^3/1,000 ft.

[29] Mean diameter was estimated for Bilby and Wasserman (1989) from their Figure 1.

1.3.2.3 Does interior unmanaged stream LWD frequency (pieces/1,000 ft) decrease as stream size increases?

No. Bilby and Ward (1989) reported that LWD frequency decreased as channel width increased (negative correlation) for westside streams. For interior streams, regression analysis reveals poor correlation of LWD frequency to stream channel width, or to stream wetted width. (See Figures B-2 through B-7). The strongest correlation found in any of these studies was for the Colorado data (Richmond and Fausch, 1995), where LWD frequency was *positively* correlated ($R^2 = 0.48$) with channel width. The Overton et al. (1995) data shows weak negative correlation ($R^2 = 0.16$).

1.3.2.4 Does interior unmanaged stream LWD diameter increase as stream size increases?

Yes, but the relationship is weak and inconsistent (see Figures B-8 through B-11). Six studies allowed examination of this relationship. The variables were not related in four of the studies, weakly related in one study, and fairly well related in one study. Regression analysis reveals marginally significant correlation coefficients for the Swan and Thompson data (Sugden, 1998 and Watson, 1997) ($R^2 = 0.06$) and Hayes (1996) ($R^2 = 0.51$).

We also attempted to examine the relationship between *effective*[30] LWD diameter and stream size. None of the study data sets available for our review reported paired *effective* LWD piece diameter and channel (or wetted) width. Relatively small wood (10 cm and sometimes smaller) acts to effectively form pools and other important habitat features in interior streams (Philbin, 1993; Potts and Anderson, 1990; Sugden, 1998); none of the studies that we reviewed identified a minimum effective diameter that was larger than the minimum qualifying LWD diameter (usually 10 cm).

1.3.2.5 Does interior unmanaged stream LWD volume decrease as stream size increases?

LWD volume has been reported to decrease as stream size increases for westside streams (Bilby and Ward, 1989; Keller et al., 1982; Marston, 1982; Swanson et al., 1982). However, we found no relationship between LWD volume and stream size (channel or wetted width) for the interior stream data sets that we examined (see Figures B-12 through B-15).

[30] Effective pieces of LWD, tallied in some studies, rather than total pieces, are generally defined as those that physically act within the channel to form pools or other habit features important for fish. LWD that provides cover is included in this definition in some studies. Acting LWD and stable LWD are also used somewhat synonymously, and we have included these terms in the effective label.

1.3.3 LWD / Stream Gradient Relationships

1.3.3.1 Does interior unmanaged stream LWD frequency increase as stream gradient increases?

Yes, although the relationship is weak. Nine of the interior study data sets allowed analysis of LWD frequency/gradient relationships. We found that LWD frequency was unrelated to gradient in three of the data sets, weakly related in one data set, and significantly related in five data sets (see Figures B-16 through B-22). We also examined the LWD frequency / gradient relationship by gradient class interval: < 1.5%, 1.5 - 4.0%, and > 4.0%. Results from each of the studies are displayed in Table B-7. We provide these gradient class results because the reviewers of these Appendix data may conclude that there is justification for choosing LWD frequency targets as a function of stream gradient (which is the only significant relationship we found between channel characteristics and LWD frequency), rather than choosing a single target for all streams. Reviewers should also note that multivariate analysis (which we have not yet conducted) might reveal important channel width, gradient, and/or stand density relationships.

1.3.3.2 Does interior unmanaged stream LWD diameter increase as stream gradient increases?

No. Six data sets allowed analysis of this relationship. LWD diameter was not related to stream gradient in any of these data sets (see Figures B-23 through B-26).

1.3.3.3 Does interior unmanaged stream LWD volume increase as stream gradient increases?

No. Six data sets allowed analysis of this relationship. LWD volume is unrelated to stream gradient in four data sets and only weakly related in two data sets (see Figures B-27 through B-30).

1.3.4 LWD / Pool Relationships

1.3.4.1 Is interior unmanaged stream LWD density (pieces/channel width) related to pool spacing or percent pool area?

No. Seven data sets allowed analysis of this relationship. LWD frequency is unrelated to pool spacing or area in five of these data sets, and only weakly related within the Thompson (Sugden, et al., 1998) and central Idaho (Overton, 1995) data sets (see Figures B-31 through B-36).

1.3.5 Managed Stream vs. Unmanaged Stream Relationships

1.3.5.1 Is interior LWD frequency lower in managed streams?

No. Six data sets allowed analysis of this relationship. Mean LWD frequency was slightly lower for some of the data sets, but only one of the data sets revealed statistically lower LWD frequency for managed streams as compared to unmanaged streams (see Tables B-8 through B-11 for managed and unmanaged LWD comparisons.)

1.3.5.2 Is interior LWD size smaller in managed streams?

No. Five data sets allowed analysis of this relationship. None of the data sets revealed statistically lower LWD size for managed streams compared to unmanaged streams.

1.3.5.3 Is interior LWD volume lower in managed streams?

Yes, but the relationship is not consistent. Five data sets allowed analysis of this relationship. Three of the data sets reveal that LWD volume in managed streams is not statistically lower than LWD volume in unmanaged streams. Bilby and Wasserman (1989) as well as Richmond and Fausch (1995) show that managed streams have lower LWD volume.

1.3.5.4 Is interior pool spacing or pool area lower in managed streams?

No. Six data sets allowed analysis of this relationship. Managed stream pool spacing or area was not different from unmanaged pool area or spacing in five of these studies. The Swan (Watson, 1997) data do indicate a significant difference in pool spacing between unmanaged and managed streams.

1.4 Summary and Conclusions

Twelve interior studies and available databases from unmanaged streams in the northwest were examined in this analysis.

Relationships between LWD and stream size, stream gradient, pool frequency, management status, and interior versus westside were examined. LWD frequency has been reported by several authors (Andrus, et al., 1988; Bilby and Ward, 1989; Bilby and Wasserman, 1989; Grette, 1985) to be correlated to density and composition of adjacent forest stands, however, stand data were not included in the data sets that we reviewed, so this relationship was not examined.

Mean interior unmanaged stream LWD frequency ranged from 25 to 169 pieces/1,000 feet, and was generally lower than LWD frequency for westside streams (range 96 to 128 pieces/1,000’). Mean interior unmanaged stream LWD diameter is generally much smaller than that measured for the three westside studies.

LWD / channel width relationships reported for westside streams are not valid for interior streams; interior LWD frequency and LWD diameter is poorly and inconsistently correlated with channel width or wetted width; interior LWD volume and stream size were not correlated.

LWD frequency in unmanaged interior streams was found to be significantly correlated with stream gradient in some of the databases examined, although LWD diameter and volume were not correlated with stream gradient.

LWD frequency in unmanaged interior streams was not found to be correlated with measures of pool area or frequency.

Small differences in unmanaged and managed stream LWD characteristics were identified in some of the data sets examined, but only volume of LWD was lower in managed versus unmanaged streams; none of the other managed/unmanaged comparisons were found to be significantly different.

Lack of correlation between LWD and stream channel characteristics suggests that valid instream LWD frequency and size goals cannot be determined as a function of channel characteristics for extrapolation to the NFHCP area. The study data examined do suggest that it is most valid for the NFHCP to derive LWD frequency and size goals from the statistics computed from the available databases. Furthermore, LWD frequency is best expressed as pieces/channel distance (i.e., pieces/1,000 feet), the measure currently used in the federal PACFISH, FEMAT, INFISH, and ICBEMP management documents.

Considering the wide variability among the studies, even among those using similar methodology, it is difficult to justify a single LWD frequency target value for the entire NFHCP area. Based on the results of the studies examined, we recommend using regional values for establishing LWD frequency targets. Eastern Washington is well represented by the Bilby and Wasserman (1989) data, with an average of 86 pieces/1,000’. For western Montana, three of the data sets used consistent methodology (10 cm minimum diameter, 2 meter minimum length) - the Swan (Watson, 1997), Thompson (Sugden, 1998), and Hayes (1996), with averages of 73, 57, and 114 pieces/1,000’, respectively. Of the three Idaho studies examined, measurement methods not only differed from those used in the other studies, but the methods also differed between the three studies themselves; this makes it impossible to derive meaningful target values for northern Idaho from these studies. We therefore recommend using the same target values for northern Idaho as for western Montana; we believe this to be justified, because northern Idaho is (geographically) more similar to western Montana than it is to eastern Washington.

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2.0 NON-FISH-BEARING STREAMS

Non-fish-bearing streams account for a large proportion of the total stream network and have the potential to influence many fish-bearing stream reaches. Where a significant proportion of flow for the fish-bearing stream comes from the non-fish bearing tributaries, significant increases in water discharge can increase bank erosion and sediment transport. Changes in coarse and fine sediment inputs to non-fish-bearing streams can fill pools and reduce the quality of spawning gravel in fish-bearing reaches. Significant reductions in large woody debris input can reduce an important roughness element that reduces hydraulic energy and traps sediment. Finally, riparian vegetation provides shade that can be important for maintaining cool water temperatures. Management strategies for non-fish-bearing streams within the HCP area have been designed primarily to maintain the important channel processes that can affect fish-bearing streams. The management strategies, however, will also contribute to many of the significant on-site ecological values of non-fish-bearing streams.

Before examining specific management strategies, the following section will describe the important channel processes that can be influenced by timber harvest, and the current scientific understanding of these processes. Then, the proposed management for non-fish-bearing streams will be considered in two parts based on whether flow is perennial or intermittent. In general, perennial non-fish-bearing streams are treated more conservatively than intermittent streams because of the greater potential for perennial streams to influence the fish-bearing portion of the stream network. Perennial streams require more protection than intermittent streams due to their higher discharges, greater sediment transport ability and greater opportunity to deliver significant amounts of energy and material directly to fish-bearing streams.

2.1 Management Objectives for LWD and Shade

Management objectives for non-fish-bearing streams that deliver stream flow, sediment and other material to fish-bearing streams will focus primarily on large woody debris (LWD) and shade. While most of the technical discussion will focus on the role of LWD, information is also provided about the role of small streams in moderating stream temperature. A more detailed discussion of small streams and stream temperature can be found in Sugden et al.(1998).

The ecological functions of LWD in non-fish bearing headwater streams include:

1. Sediment storage in deposits upstream of debris jams that buffers delivery of sediment eroded from hillslopes to downstream fish-bearing reaches;
2. Flow resistance that may reduce sediment transport capacity and slow the rate of delivery of peak flows;
3. Habitat (e.g. pool formation, gravel storage and organic substrate) for amphibians, invertebrates, and micro-organisms;
4. Nutrients for aquatic organisms on-site (e.g., amphibians, invertebrates, and micro-organisms) and downstream (e.g., fish) and;

Water quality (e.g. water temperature)

Although there is a growing scientific literature documenting aspects of these functions, particularly in coastal forest ecosystems, regional variation of the systems and processes are poorly understood (e.g. TFW 1996). Enough is known to conclude that LWD is a potentially significant component of the aquatic ecosystem, but not enough to describe specific sensitivities that would allow "tailoring" of management actions to the degree that is possible for fish-bearing streams. The scope of the proposed HCP, however, restricts management concerns primarily to potential downstream effects on fish-bearing streams, which reduces the impact of scientific uncertainty on management prescriptions.

The role of LWD in sediment routing is the primary channel process that has been subjected to quantitative analysis. The examination of potential management impacts on non-fish-bearing streams will focus on sediment routing and assume that functional targets designed to address sediment routing will be adequate for other less-well-quantified ecosystem functions. Owing to the relative scarcity of data regarding LWD function in smaller, steeper streams, it is necessary to assess existing data sets for applicability to the project area. The following sections will examine "natural" LWD load and function for these streams, assess the evidence for management effects on LWD load and function, and address potential targets to evaluate management strategies.

2.2 Background

With respect to LWD load in small streams and potential downstream effects, several authors have suggested that channel obstructions (LWD jams or steps) have a significant influence on sediment storage and routing (Bilby and Likens 1980, Mosely 1981 Megahan 1982, Sidle 1988, MacDonald and Ritland 1989). Potential downstream effects of sediment routing are a complex function of stream transport capacity, sediment supply, and in-stream storage. O'Connor (1994) and O'Connor and Harr (1994) modeled this sediment storage and routing and the potential influence of timber harvest on the abundance of LWD steps based on field observations in the northwest Olympic Mountains of western Washington. This study suggested that natural decay of LWD steps over a model interval of 60 years, with no new step formation resulting from a lack of LWD recruitment from riparian forest stands, could lead to significant increases in average bedload sediment yield ranging from about 40 to 120%. These rates of increased bedload yield resulted from decay of about half of the original LWD steps over the 60-year modeling interval.

LWD steps are thought to be significant because they temporarily immobilize sediment in stream channels that are otherwise supply-limited with respect to sediment routing. Sediment storage in small steep stream is expected to be low owing to high transport capacity, and LWD steps have been shown to store the equivalent of decades of bedload sediment yield (Megahan 1982, O'Connor 1994). Potential downstream effects (e.g., channel sedimentation), however, have not been demonstrated. This requires watershed-level sediment budget studies that quantify the relative magnitude of sediment storage and transport and sediment sources and evaluate the potential ecological consequences of increased sediment of the specified magnitude. Until such studies are undertaken, it is necessary to assume that there are*potential* deleterious effects that could result from a significant decline in LWD and LWD steps in small streams. With respect to other ecological functions of LWD, it is assumed that adequate function will be maintained provided that LWD load (or other measure of LWD abundance) is maintained within the natural range.

2.3 Abundance of LWD Steps

Previous studies have suggested that the frequency or spacing of boulder steps in steep streams is a function of channel slope (e.g. Whittaker and Jaeggi 1982, Abrahams et al. 1995). Data for LWD steps collected from published and unpublished sources from various regions in western North America indicate that there is a significant relationship between LWD step abundance and channel slope (Figure B-37). These data include both managed and unmanaged sites. (Inferences regarding management effects on observed LWD step abundance are explored using a subset of the data for a region where the number of managed and unmanaged sites is reasonably comparable).

Two conclusions may be drawn from these data. First, there is a positive relationship between LWD step abundance and stream slope, with wide scatter in the data, indicating that steeper streams tend to have more LWD steps. Second, although there may be a hint of some regional differences, given the scatter of data, it is reasonable to hypothesize that there are no significant differences in LWD step abundance among regions. This assumption allows available data to be extrapolated to the project area. In the context of channel geomorphology and classification, these data could be interpreted to present potential targets for different channel types based on slope class (Table B-12).

Perennial channels with gradients <6% are likely to be fish-bearing, consequently the relevant channel type for this discussion is the cascade type (Table B-12). A LWD step target could be generalized to this level, however, consideration would need to be given to the data scatter, that is, the wide variance in the range of normal. An alternative application of the data that could be used to develop somewhat more sophisticated targets is presented in Table B-13, where LWD step abundance and spacing are predicted using the regression relationship derived from Figure B-37 (LWD abundance (#/100m) = 0.41 x gradient (%)).

2.4 LWD Step Height and Sediment Storage Capacity

Further evidence suggesting that characteristics of LWD steps in steep streams are fundamentally independent of region is presented in Figure B-38 and Table B-14. These data for step heights in different regions include both managed and unmanaged streams. Figure B-38 indicates that unlike the relationship between step abundance and gradient, there is no relationship between step height and gradient. This is particularly relevant in that step height is a good predictor of sediment storage associated with a step. The statistical characteristics of step height are best described by a central tendency of observations.

Table B-14 summarizes observations from 4 regions (the 3 regions in Figure B-38 and Montana). It is evident that the median of the distribution of step heights is regionally invariant at about 0.6 to 0.7 m. The average values are somewhat more variable and reflect the influence of relatively rare steps > 1.0 m high. Although the heights in areas that are in or representative of the HCP project area

(Cascades and northern Rocky Mountains) are somewhat smaller than in the Olympic Mountains of western Washington, the differences are not dramatic.

2.5 Influence of Step Height on Sediment Storage

Step height is an estimate of sediment storage in LWD steps. Table B-15 summarizes regression analyses of the relationship between step height and sediment storage volume per unit bankfull channel width. The relationships for the Cascades and Montana are likely to be more representative of the project area. The Olympics data suggest negative storage in steps up to 0.5 m height, likely reflecting the larger sediment volumes found in that region caused by relatively high rates of mass wasting and sediment transport. This suggests that a more complex regression model may be warranted for the Olympic data. The influence of step height on storage volumes is the basis for arguing that a variety of step heights (and therefore LWD diameters, including the larger diameters) is functionally significant.

A key distinction between LWD steps and other sediment storage, such as bars formed in the lee of LWD obstructions, boulders or boulder steps, is that they can create sediment storage reservoirs with a depth greater than the scour depth of the stream bed, thereby creating a storage element that hides sediment from the hydraulic forces of the stream until the LWD step fails. Other storage elements have depths less than or equal to the scour depth of the channel, and therefore typically add to capacity of the active streambed to store sediment.

In the northwest Olympics, O’Connor (1994) found average scour depth for typical flow events over the entire stream bed to be about 0.1 m. In sediment deposits formed above LWD steps, however, the scour depth was greater (as high as 0.35 m). It is assumed that scour in the sediment deposit upstream of steps can typically extend to about 0.3 m depth. O’Connor’s (1994) simulation model assumed (on the basis of field observations) that about 2/3 of sediment storage upstream of a LWD step was in this deeper storage compartment, and the remainder was in the active surface compartment. This value may well vary among regions on the basis of hydrology and its controlling influence on sediment transport.

The point here is that the deeper storage is more permanent, and has a different effect when considering sediment routing. This type of storage can be viewed as a source of sediment in a system of declining LWD and as a sink in a system of increasing LWD. The time scale of routing for these sediments in these systems is probably decades. These data and interpretations support the case for a riparian management strategy that provides a variety of LWD diameters and some larger step heights. Note that step height does not equate with LWD diameter forming the step in all cases. Arrangements of multiple LWD pieces and boulder foundations are common. It is less typical to find a solitary 1-m diameter log forming a step.

Major unknowns include the process of LWD weathering, breakage, decay and transport that transforms LWD that is recruited to the vicinity of a stream channel into complex features of stream morphology (steps) that store sediment relatively efficiently and for relatively long periods of time. In addition, the occurrence of LWD steps and sediment storage in headwater streams in the HCP project area is not well documented. However, given the foregoing evidence suggesting that LWD step characteristics are not significantly different among regions, it is possible to assess one relatively detailed data set that provides insights regarding the relationship between in-stream LWD, LWD steps and forest management.

2.6 Relationship Between LWD Abundance and Step Abundance

Data collected by O'Connor (unpublished) during a watershed analysis project in the southeastern Olympic Mountains of western Washington shows no functional relationship between LWD abundance and LWD step abundance (Figure B-39, Table B-16). One conventional measure of LWD abundance standardizes the number of pieces by the bankfull channel width. It was hypothesized that the abundance of LWD steps would be a simple function of LWD abundance; this was not the case for these data (Figure B-39). Based on central tendencies (mean and median) for the data set, there was about one LWD step for every ten pieces of LWD.

Moreover, there was no statistically significant (paired t-test, one-tailed, $p<0.05$) difference between LWD abundance of LWD step abundance between unmanaged streams and managed streams (Table B-16). The data suggest a tendency for managed streams to have comparable or slightly greater LWD abundance and to have somewhat lower LWD step abundance.

2.7 Management Considerations

Given previous studies that indicate LWD turnover time of several decades (Lienkaemper and Swanson 1987) and decay rates of a few percent per year (Murphy and Koski 1989), it is likely that short term declines in LWD abundance are likely to be subtle or undetectable (as suggested by the data above), provided that overt LWD removal does not occur. LWD recruitment in second-growth forest stands begins in about 25 to 30 years for hardwoods and about 50 to 60 years for conifers (Andrus et al. 1988, Grette 1985). Thus, it may be hypothesized that logging riparian stands completely may suppress recruitment for about 50 to 60 years from coniferous forests, and that in-stream abundance may not be reduced for a period of several decades (Hedin et al. 1988). Therefore, forest management impacts related to harvest in riparian forest stands on LWD load in streams can be expected to persist for a period of decades adjacent to clear cut forest stands, beginning a few decades after initial harvest, or sooner if LWD was removed from streams. On the other hand, if continuous inputs of LWD are maintained over time, it is likely that LWD function would be maintained at a level sufficient to render downstream impacts insignificant and undetectable.

While the amount of LWD input is a critical factor, the amount of time required to process LWD is an equally important factor. The relatively low ratio of LWD pieces to highly-functional LWD (i.e., steps) indicated in Figure 3 and Table 5 is consistent with findings of O'Connor and Ziemer (1989) regarding the proportion of LWD recruited to a valley bottom that is highly-functional as an integrated component of stream morphology (i.e., associated with pools). The low ratio may be largely attributable to random events such as location of fallen trees in relation to the streambed. However, the degree of breakage and decay of LWD influences the likelihood of a particular fallen log becoming well-integrated in the streambed, suggesting that time may be a significant factor. If relatively slow or infrequent stream processes that contribute to positioning of LWD (bank erosion, channel migration, and high-magnitude floods), are taken into account, then processing time for LWD is a likely to be a significant element of the complex processes responsible for LWD function in streams.

2.8 Small Streams and Water Temperature in Downstream Fish-Bearing Waters

The principal source of heat energy for small streams is solar radiation striking the surface of the stream (Brown 1969). The amount of sunlight that reaches the stream depends on the stream surface area and the shading provided by vegetation and topography (Brown and Krygier 1970). Vegetation (trees and brush) can provide substantial shade to small streams. Shade is typically approximated through canopy closure measurements.

The magnitude of potential temperature change varies with stream size and elevation. Smaller streams will both heat and cool more quickly without shade than larger streams (Adams and Sullivan 1990).

Smaller streams have limited potential, however, to increase downstream water temperature in fish-bearing streams (Caldwell et al 1991). These small perennial streams typically have a large proportion of groundwater inputs which tends to be very cool. Groundwater temperature typically mirrors soil temperature which is typically less than 60 degrees F. These streams are thus typically cooler than downstream receiving waters. The small streams also have limited ability to influence receiving waters unless it provides a significant proportion (more than 20%) of flow at the confluence (Caldwell et al. 1991).

While limited data exist on the temperature regimes of small streams, some research indicates that stream temperatures can quickly equilibrate to current shading levels within 500 feet of stream length (Caldwell et al. 1991). Where small streams provide more than 20% of the total flow to a fish-bearing stream, leaving adequate shade for 500 feet upstream of its confluence should maintain cool water temperatures.

2.9 References

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Table B-1. Features of studies that surveyed LWD characteristics in streams flowing through unmanaged forests of the Interior Columbia River Basin, N. central Colorado, and west of the Cascade Mountain crest.

Eastside Studies										
Author	**Location / drainage**	**Average stream Width (ft)**	**Bankfull or wetted width**	**# of streams surveyed**	**# of reaches surveyed**	**Managed or unmngd.**	**# of unmngd reaches surveyed**	**Total or acting LWD**	**Minimum LWD definition**	
									diameter (cm)	**length (m)**
Watson	Swan	20.5	bankfull		52	both	20	total	10	2
Sugden	Thompson	15.7	bankfull		107	both	76	total	10	2
	Sw&Thom.	16.7	bankfull		159	both	96	total	10	2
Richmond	Colorado	17.6	bankfull	11	15	both	11	total	10	2 †
USFS	Bitterroot	21.5		50	159	unmngd	159		15	1
Overton	central Idaho	23.3	wetted	62	218	unmngd	218	total *	10	~2.8
Bilby	E. Wash.	33.3	bankfull	51	51	both	29	total	10	2
USFS	Lolo		wetted	69	69	both	25	acting	10	2/3 width
Hayes	Montana	19.5	bankfull	8	40	unmngd	40	total	10	2
Potts	Montana	6.5	bankfull	1	8	unmangd	8	total	10	
Huntington	N. Idaho		wetted	250	1,302	both	547	acting	10	
Knight	E. Oregon	14.2	bankfull	13	13	both	7	total	10	1
Philbin	N. Idaho	< 10	bankfull	80	80	unmngd	80	total *	10	
Westside Studies										
Bilby	W. Wash.	31.7	bankfull	22	22	unmngd	80	total *	10	2
Fausch	Br. Columbia	7.9	bankfull	1	4	unmngd	4	total	10	1

* — Modified total piece count: only some pieces within aggregations were reported, with exact methods varying by study.
† — Original data was gathered for minimum piece length of 1m; we adjusted to 2m minimum based on length distributions given in the paper.

Table B-2. LWD Relations in Interior Streams.

Study area	**Swan**	**Thompson**	**Sw&Thom.**	**Colorado**	**Bitterroot**	**Cen. Idaho**
Author	**Watson**	**Sugden**		**Richmond**	**USFS**	**Overton**
LWD frequency lower in larger streams	N	N	N	N	N	W
LWD size larger in larger streams	N	W	W	N		
LWD volume lower in larger streams	N	N	N	N		
LWD frequency higher in steeper streams	Y	W	W	N	N	Y
LWD size larger in steeper streams	N	N	N	N		
LWD volume higher in steeper streams	W	W	W	N		
LWD density related to pool spacing (or area)	N	W	W	N	N	W
LWD frequency lower in managed streams	N	N	N	Y		
LWD size smaller in managed streams	N	N	N	N		
LWD volume lower in managed streams	N	N	N	Y		
Pool spacing (or area) lower in managed streams	Y	N	N	N		

Study area	**E. Wash.**	**Lolo**	**Montana**	**N. Idaho**	**E. Oregon**	**Potts**	**Overall**
Author	**Bilby**	**USFS**	**Hayes**	**Huntington**	**Knight**	**Montana**	**Conclusion**
LWD frequency lower in larger streams	N		N				N
LWD size larger in larger streams	N		Y		N		N
LWD volume lower in larger streams	N		N		N		N
LWD frequency higher in steeper streams	N		Y	Y		Y	W
LWD size larger in steeper streams	N		N		N		N
LWD volume higher in steeper streams	N		N		N		N
LWD density related to pool spacing (or area)				N	N		N
LWD frequency lower in managed streams	N	N		N			N
LWD size smaller in managed streams	N				N		N
LWD volume lower in managed streams	Y				N		W
Pool spacing (or area) lower in managed streams		N		N	N		N

Y = study supports conclusion; N = study does not support conclusion; W = study weakly supports conclusion

Table B-3. LWD loading in unmanaged watersheds (measured in number of pieces/1,000').

Author	Location/ drainage	# of reaches surveyed	Average LWD frequency	Standard deviation	Standard error of the mean	95% confidence limits
Eastside Studies						
Watson	Swan	20	73.2	58.5	13.1	27.3
Sugden	Thompson	76	57.2	38.8	4.5	8.9
	Sw&Thom.	96	60.5	43.8	4.5	8.9
Richmond	Colorado	11	85.9	30.7	9.3	20.6
USFS	Bitterroot	159	48.2	33.3	2.6	5.2
Overton	Central Idaho	218	24.6	21.4	1.4	2.8
Bilby	E. Wash.	29	86.1	44.1	8.2	16.8
USFS	Lolo	69	118.0	*	*	*
Hayes	Montana	40	113.8	63.9	22.6	53.3
Potts	Montana	8	36.7	19.0	6.7	15.9
Huntington	N. Idaho	547	43.8	38.6	1.7	3.2
Knight	E. Oregon	*	*	*	*	*
Philbin	N. Idaho	80	169.4	*	*	*
Westside Studies						
Bilby	W. Wash.	22	97.6	61.9	13.2	27.5
Harmon	SE Alaska	*	96.0	*	*	*
Fausch	Br. Columbia	4	128.0	42.5	21.3	67.6

Table B-4. LWD density in unmanaged watersheds (measured in number of pieces/stream width)

Eastside Studies						
Author	**Location/ drainage**	**# of reaches surveyed**	**Average LWD frequency**	**Standard deviation**	**Standard error of the mean**	**95% confidence limits**
Watson	Swan	20	1.73	1.83	0.41	0.86
Sugden	Thompson	76	0.95	1.13	0.13	0.26
	Sw&Thom.	96	1.11	1.34	0.14	0.27
Richmond	Colorado	11	1.65	1.19	0.36	0.80
USFS	Bitterroot	159	0.94	1.06	0.08	0.16
Overton	Central Idaho	218	0.47	0.34	0.02	0.05
Bilby	E. Wash.	26	2.64	1.81	0.35	0.73
USFS	Lolo	*	*	*	*	*
Hayes	Montana	40	2.16	1.58	0.56	1.32
Potts	Montana	*	*	*	*	*
Huntington	N. Idaho	*	*	*	*	*
Knight	E. Oregon	*	*	*	*	*
Philbin	N. Idaho	*	*	*	*	*

* — requires obtaining original data set

Table B-5. LWD size in unmanaged watersheds (midpoint diameter in inches).

Author	Location/ drainage	# of reaches surveyed	Average LWD frequency	Standard deviation	Standard error of the mean	95% confidence limits
Eastside Studies						
Watson	Swan	18	7.9	58.5	13.1	27.3
Sugden	Thompson	74	9.2	38.8	4.5	8.9
	Sw&Thom.	92	8.9	43.8	4.5	8.9
Richmond	Colorado	11	7.7	30.7	9.3	20.6
USFS	Bitterroot					
Overton	Central Idaho					
Bilby	E. Wash.	29	11.3	3.0	0.6	1.2
USFS	Lolo					
Hayes	Montana	40	8.5	2.0	0.7	1.7
Potts	Montana					
Huntington	N. Idaho					
Knight	E. Oregon	7	9.7	1.5	0.6	1.4
Philbin	N. Idaho					
Westside Studies						
Bilby	W. Wash.	22	16.0	*	*	*
Harmon	SE Alaska	*	20.9	*	*	*
Fausch	Br. Columbia	4	10.5	2.2	1.1	3.5

* — requires obtaining original data set

Table B-6. LWD volume in unmanaged watersheds (measured in ft^3/1,000').

Author	Location/ drainage	# of reaches surveyed	Average LWD frequency	Standard deviation	Standard error of the mean	95% confidence limits
Eastside Studies						
Watson	Swan	20	572	649	145	303
Sugden	Thompson	76	599	568	65	130
	Sw&Thom.	96	593	582	59	118
Richmond	Colorado	11	931	407	123	274
USFS	Bitterroot					
Overton	Central Idaho					
Bilby	E. Wash.	29	141	123	23	47
USFS	Lolo					
Hayes	Montana	40	870	636	225	531
Potts	Montana					
Huntington	N. Idaho					
Knight	E. Oregon	7	2,340	996	376	922
Philbin	N. Idaho					
Westside Studies						
Bilby	W. Wash.	*	*	*	*	*
Harmon	SE Alaska	*	6,240	*	*	*
Fausch	Br. Columbia	4	4,650	3,270	1,635	5,199

* — requires obtaining original data set

Table B-7. LWD Frequency by Gradient Class

Author	Location/ drainage	LWD (pieces/1,000')			Significantly different?
		< 1.5%	1.5%-4%	> 4%	
Watson	Swan & Thompson	37.9	60.3	92.8	Y
Richmond	Colorado	86.2	84.2	99.1	N
USFS	Bitterroot	36.0	44.6	49.5	N
Overton	Central Idaho	18.4	24.5	31.5	Y
Bilby	E. Wash	---	83.9	82.0	N
Hayes	Montana	---	74.6	137.3	N †
Huntington	N. Idaho	41.5	30.8	50.0	*

† — difference was not significant due to small sample size; however, linear regression analysis indicated a significant positive relationship (see Figure 5-5).

* — requires obtaining original data

MANAGED/UNMANAGED COMPARISONS.

Table B-8. LWD frequency (pieces/1/000')

Location	Managed	Unmanaged	Significantly different?
Swan&Thompson	69.1	60.5	N
Colorado	32.7	85.9	Y
E. Wash	83.4	86.1	N
Lolo	97.3	117.5	N
N. Idaho	45.5	43.8	N

Table B-9. LWD diameter (in)

Location	Managed	Unmanaged	Significantly different?
Swan&Thompson	8.7	8.9	N
Colorado	7.4	7.7	N
E. Wash	10.1	11.3	N
E. Oregon	10.9	9.7	N

Table B-10. LWD volume (ft^3/1,000')

Location	Managed	Unmanaged	Significantly different?
Swan&Thompson	677	593	N
Colorado	203	931	Y
E. Wash	79	141	Y
E. Oregon	2,980	2,340	N

Table B-11. Pool frequency (Swan & Thompson in channel widths/pool, others in % pool area)

Location	Managed	Unmanaged	Significantly different?
Swan&Thompson	12.7	11.7	N
Colorado (wd/pool)	6.6	5.5	N
Colorado (%)	9.8%	17.8%	N
Lolo	17.6%	18.6%	N
N. Idaho	23.7%	22.9%	N
E. Oregon	19.5%	16.6%	N

Table B-12. Observed LWD Step Frequency According to Channel Morphologic Gradient Class

Morphology	Gradient Range (%)	# of Reach Observations	Mean LWD Steps/100 m
Pool-riffle	s =1.5	8	0.2
Plane-bed / Forced Pool Riffle	1.5 < s = 3	16	0.5
Step-pool	3 < s = 6.5	27	1.5
Cascade	6.5 < s = 20	32	6.7
Cascade +	s >20	20	10.4

Table B-13. Predicted mean LWD step abundance and spacing derived from Figure B-37.

Channel Gradient (%)	LWD Steps per 100m	Mean Step Spacing (m)
6	2.5	41
8	3.3	30
10	4.1	24
12	4.9	20
14	5.7	17
16	6.6	15
18	7.4	14
20	8.2	12

Table B-14. Observed LWD Step Height (m) in Four Mountain Regions of Western U.S.

Distribution Parameter	NW Olympics	SE Olympics	Eastern Cascades	Montana
N	99	124	25	26
Median	0.7	0.7	0.6	0.6
Mean	--	0.87	0.59	0.77
Std. Dev.	--	0.49	0.30	0.46
Std. Error	--	0.04	0.06	0.09

Table B-15. Step height influence on sediment storage by LWD steps.

Regression Parameter	SE Olympics	Eastern Cascades	Montana
coefficient	8.29	1.71	4.69
constant	-4.49	-0.25	-1.68
r-square	0.61	0.54	0.90
n	122	25	26
Predicted Sediment Storage per Unit Channel Width (m^3/m) as a Function of LWD Step Height			
h=0.3 m	-2.00	0.26	-0.27
h=0.4 m	-1.17	0.43	0.20
h=0.5 m	-0.35	0.61	0.67
h=0.6 m	0.48	0.78	1.13
h=0.7 m	1.31	0.95	1.60
h=0.8 m	2.14	1.12	2.07
h=0.9 m	2.97	1.29	2.54
h=1.0 m	3.80	1.46	3.01
h=1.5 m	7.95	2.32	5.36

Table B-16. Mean and median LWD abundance and LWD step abundance expressed as LWD pieces (or steps) per unit channel length expressed as bankfull width (BW) for both managed (logged riparian forest) and unmanaged (old-growth riparian forest).

	Unmanaged		**Managed**	
	LWD/BW	**Steps/BW**	**LWD/BW**	**Steps/BW**
Median	1.6	0.28	2.4	0.17
Mean	2.8	0.34	2.8	0.21

Figure B-1. Regional comparison of large woody debris (LWD) frequencies in streams flowing through unmanaged riparian forests of the Interior Columbia River Basin and north-central Colorado. *Note that survey methods were dissimilar among studies, and therefore results may not be directly comparable. See text for details.*

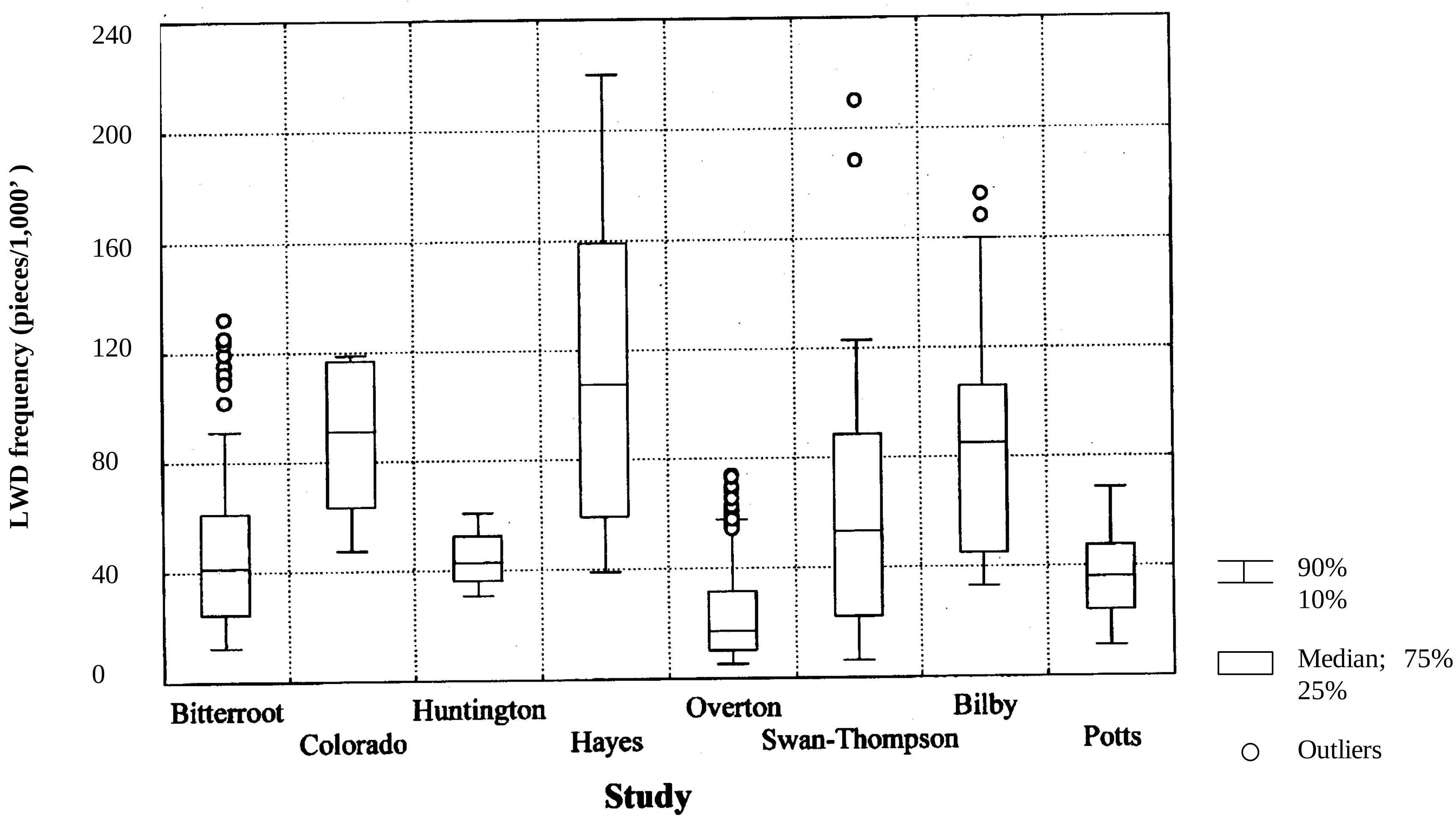


Figure B-2. Swan & Thompson Data. Undisturbed only.

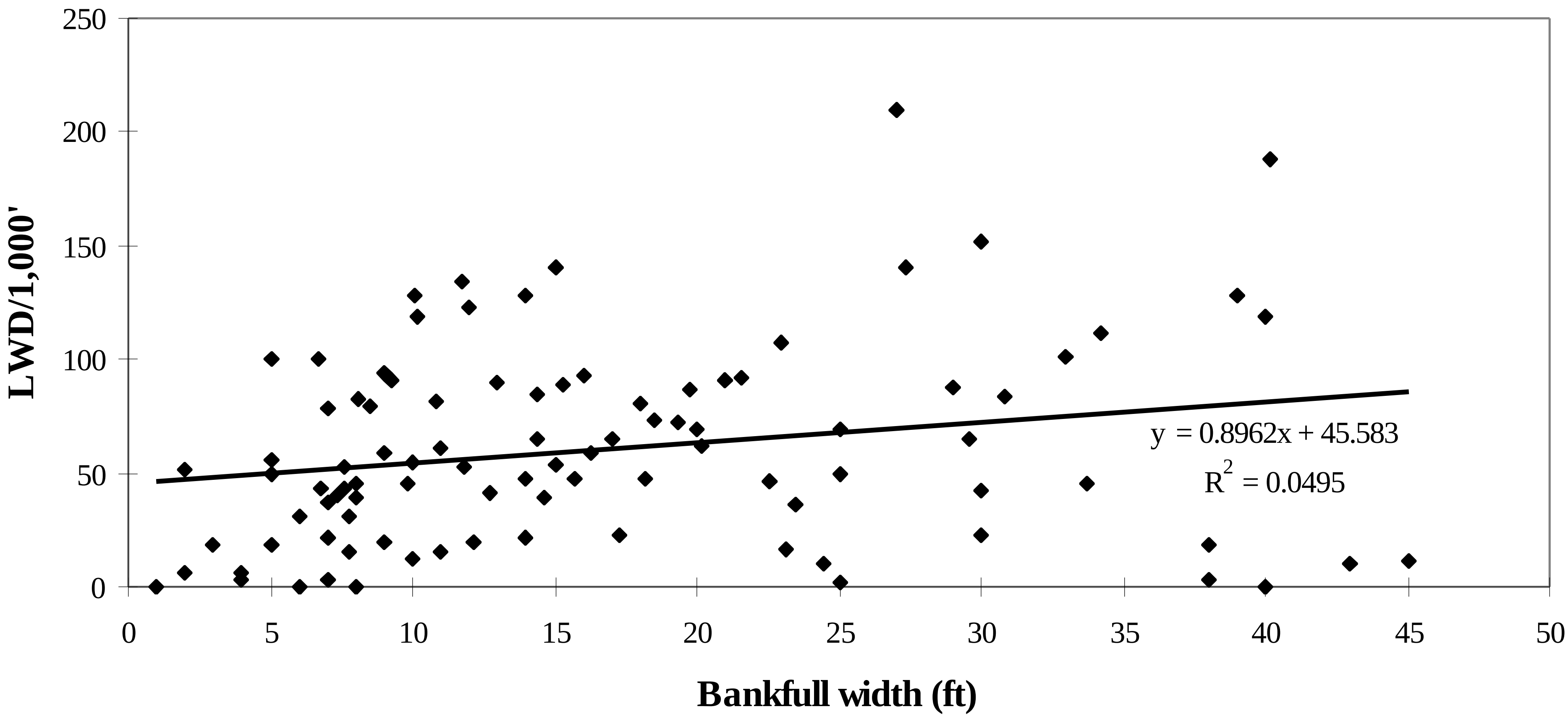


Figure B-3. Colorado Unmanaged Data.

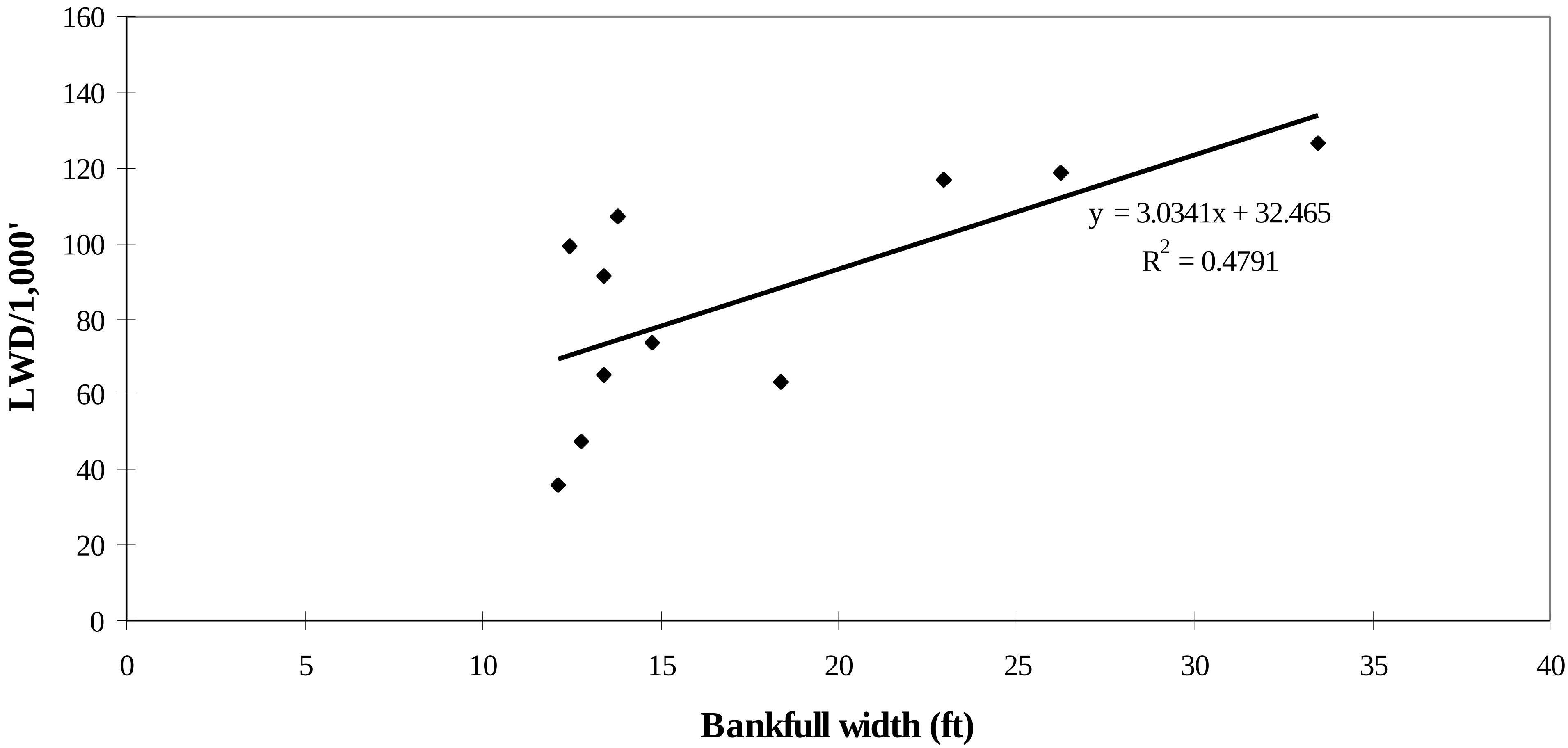


Figure B-4. Bitterroot National Forest

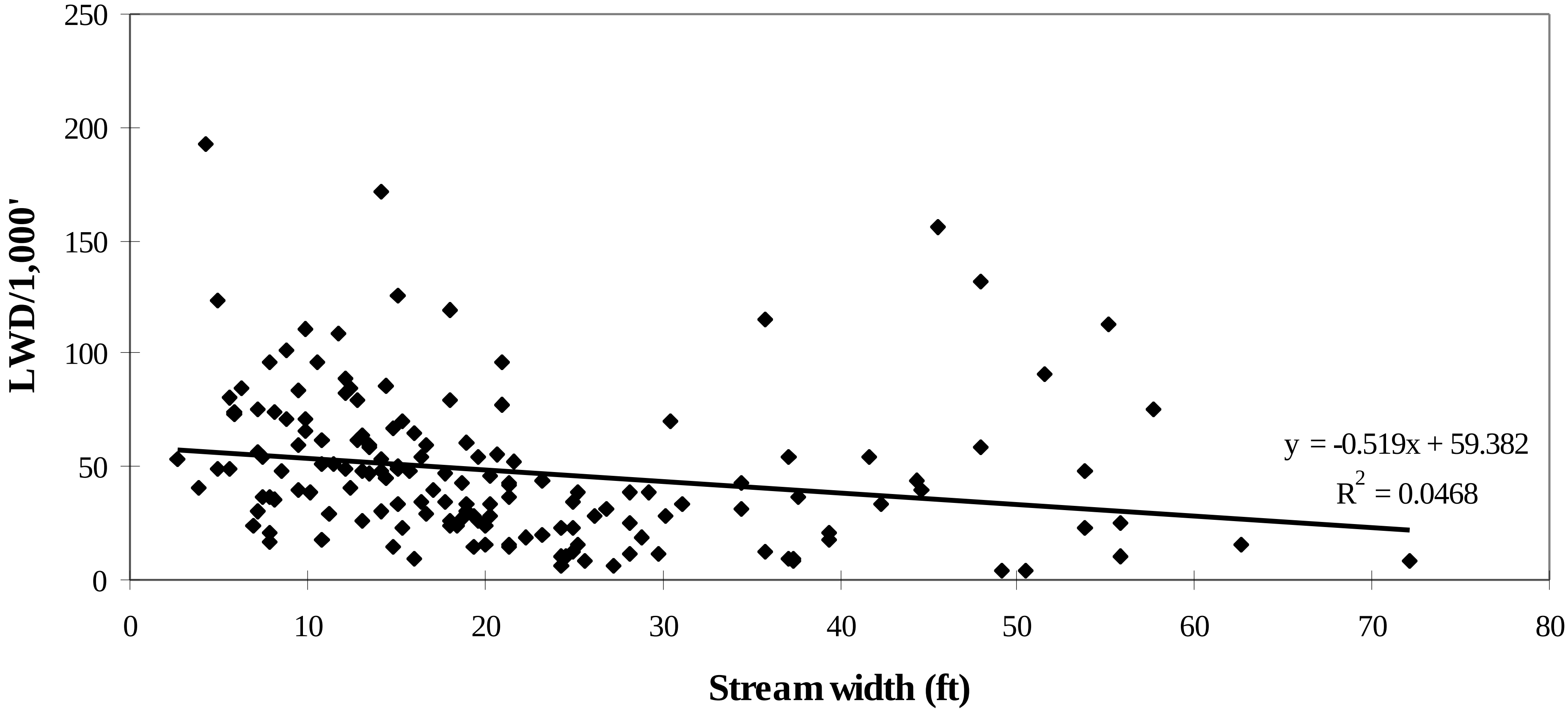


Figure B-5. Idaho (Overton)

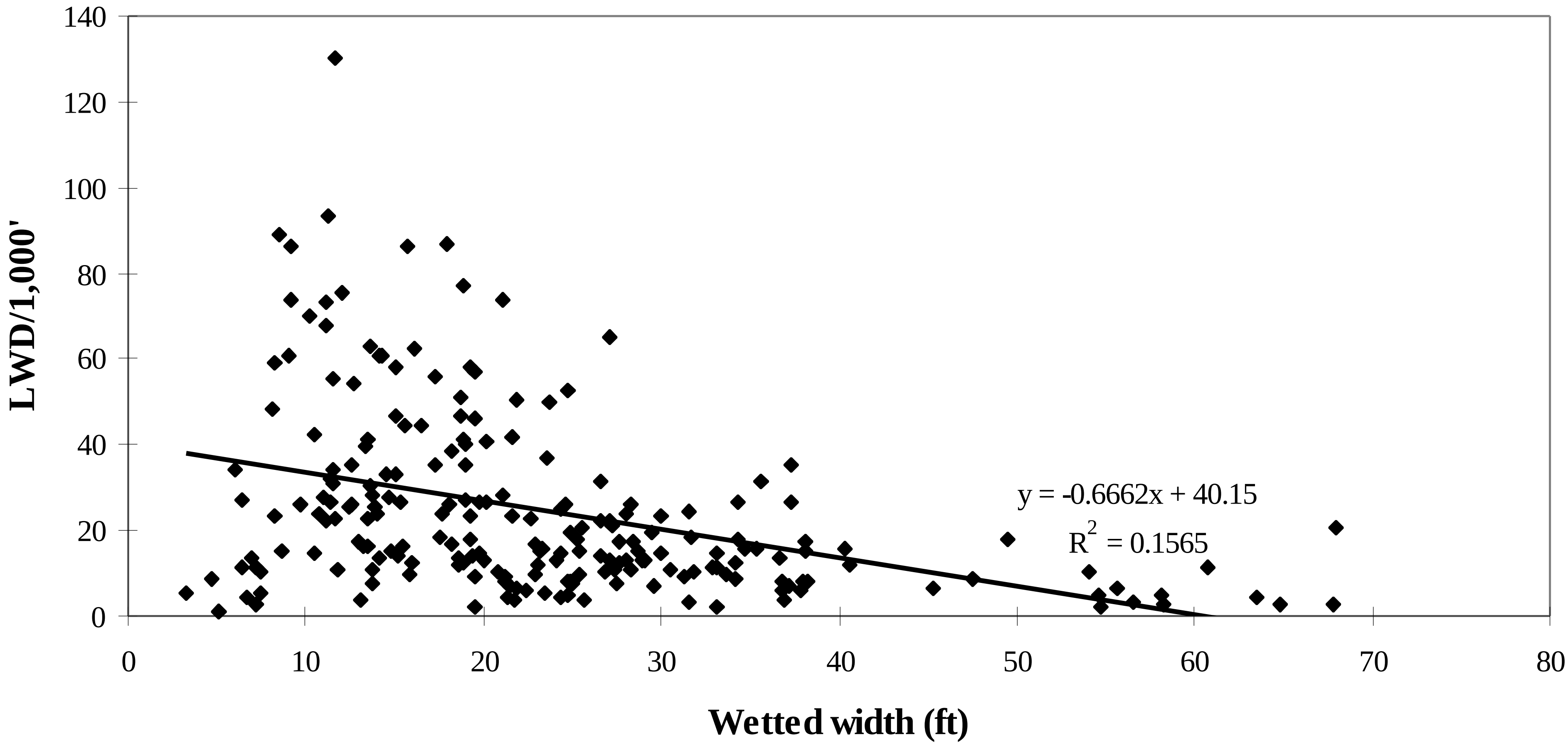


Figure B-6. Bilby — E. Washington

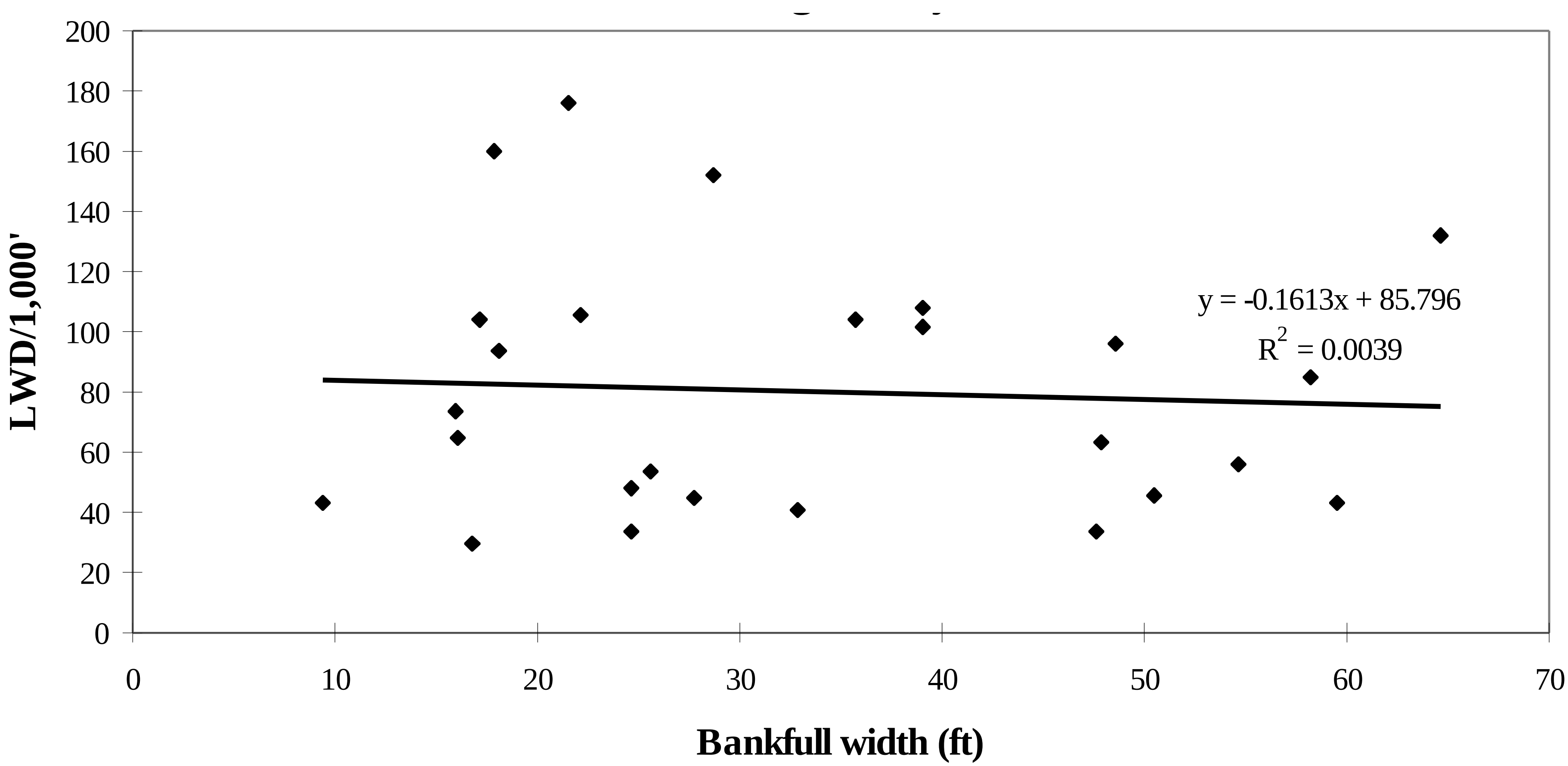


Figure B-7. Montana-Hayes

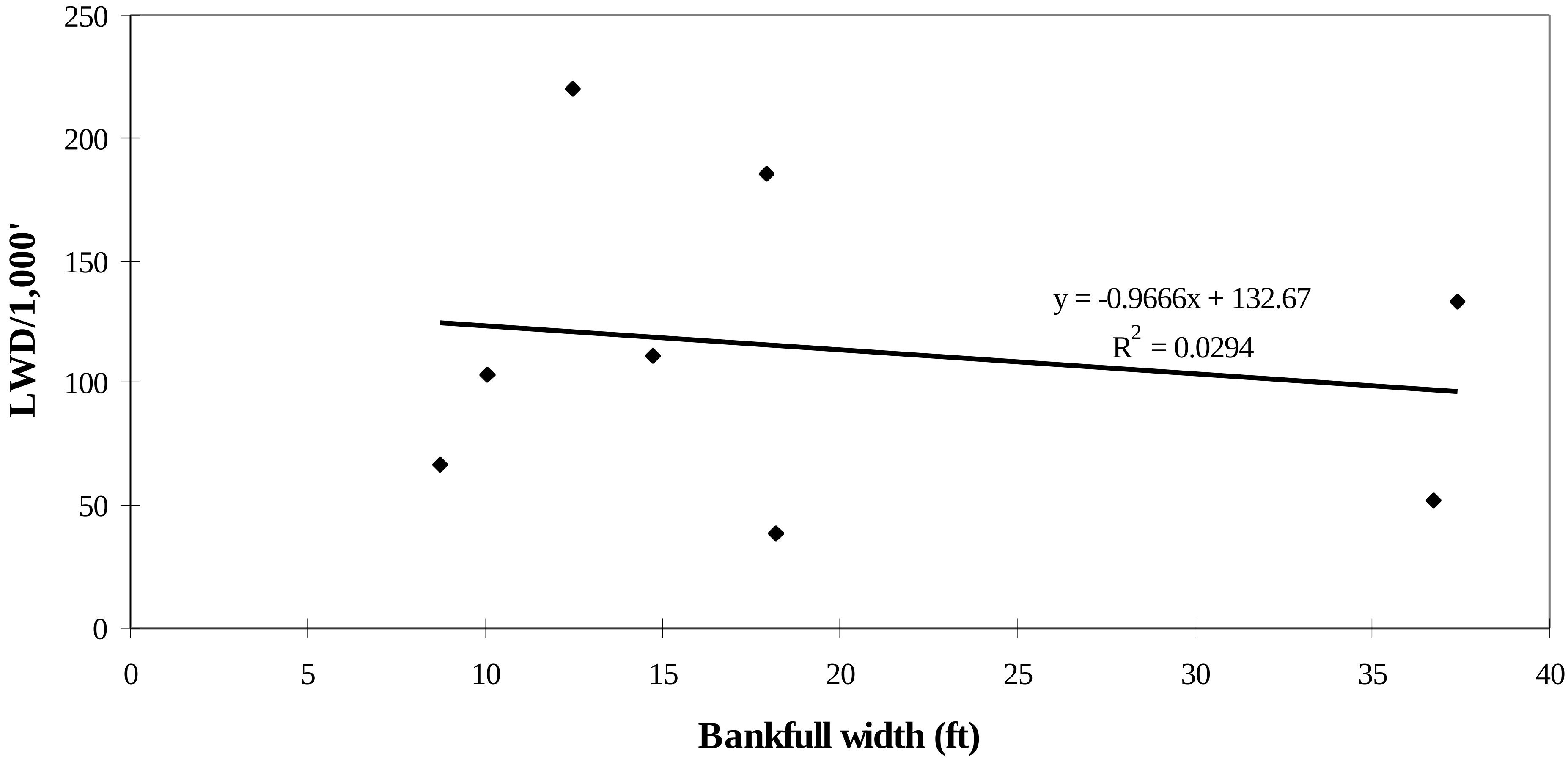


Figure B-8. Swan & Thompson Data. Undisturbed only.

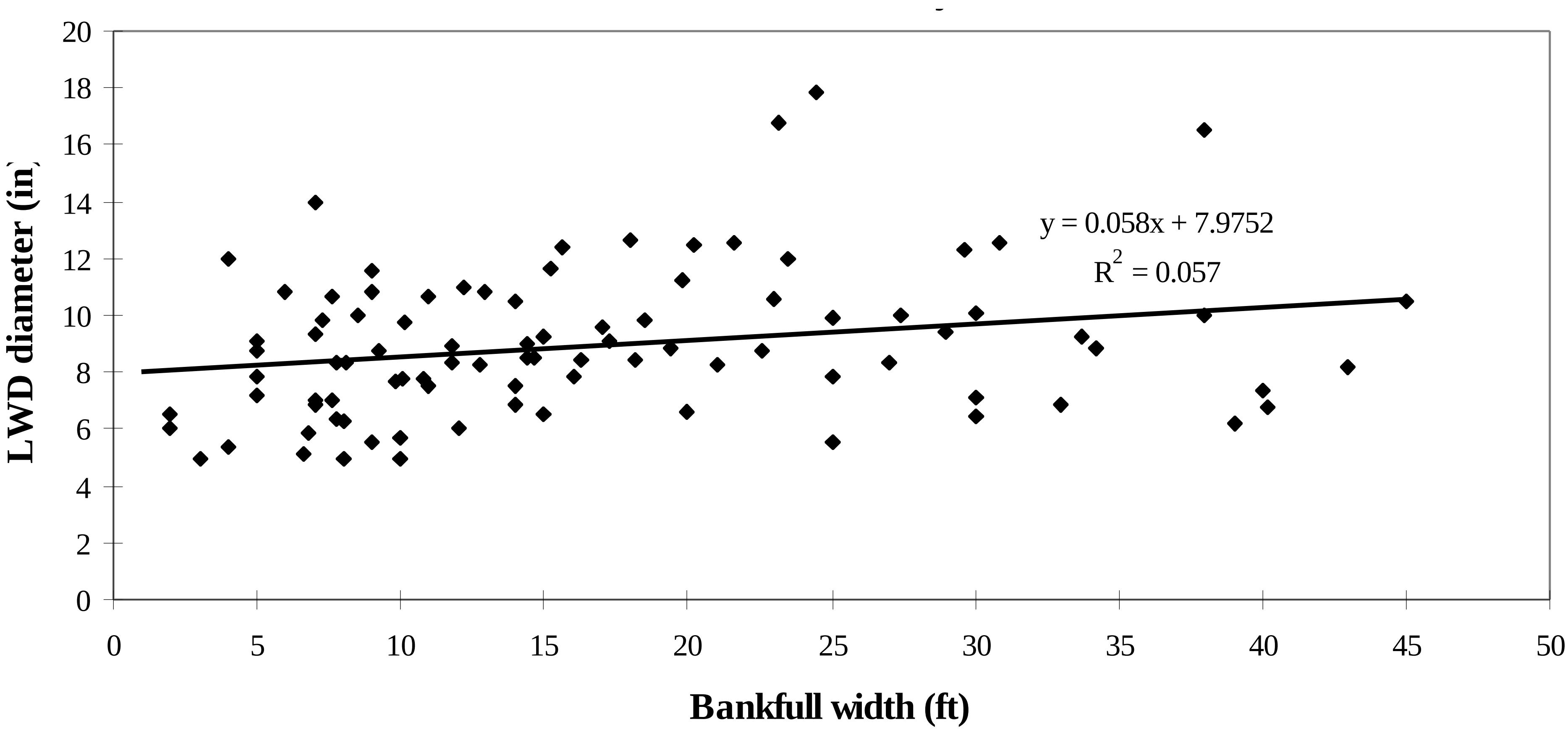


Figure B-9. Bilby — E. Washington. Unmanaged only.

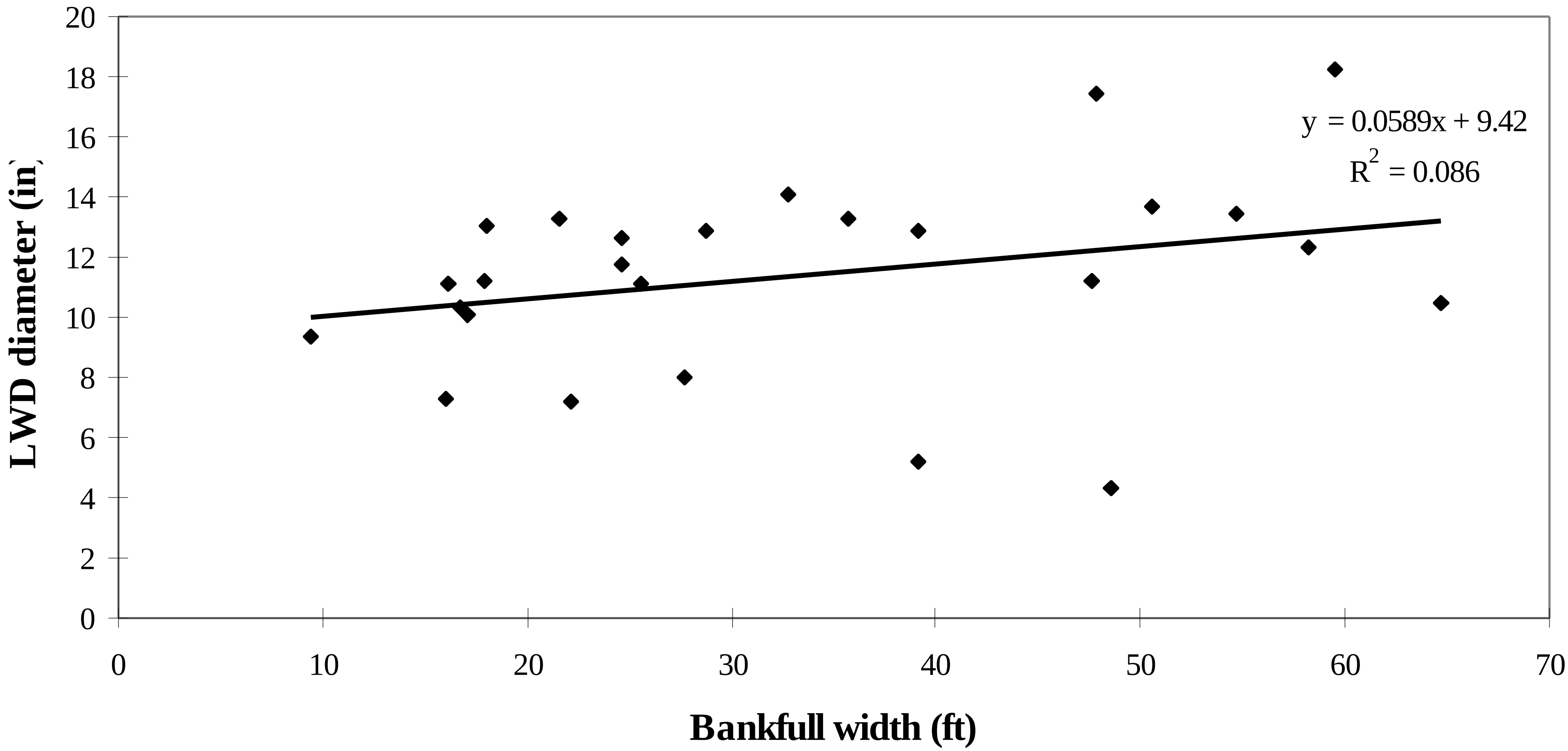


Figure B-10. Montana – Hayes.

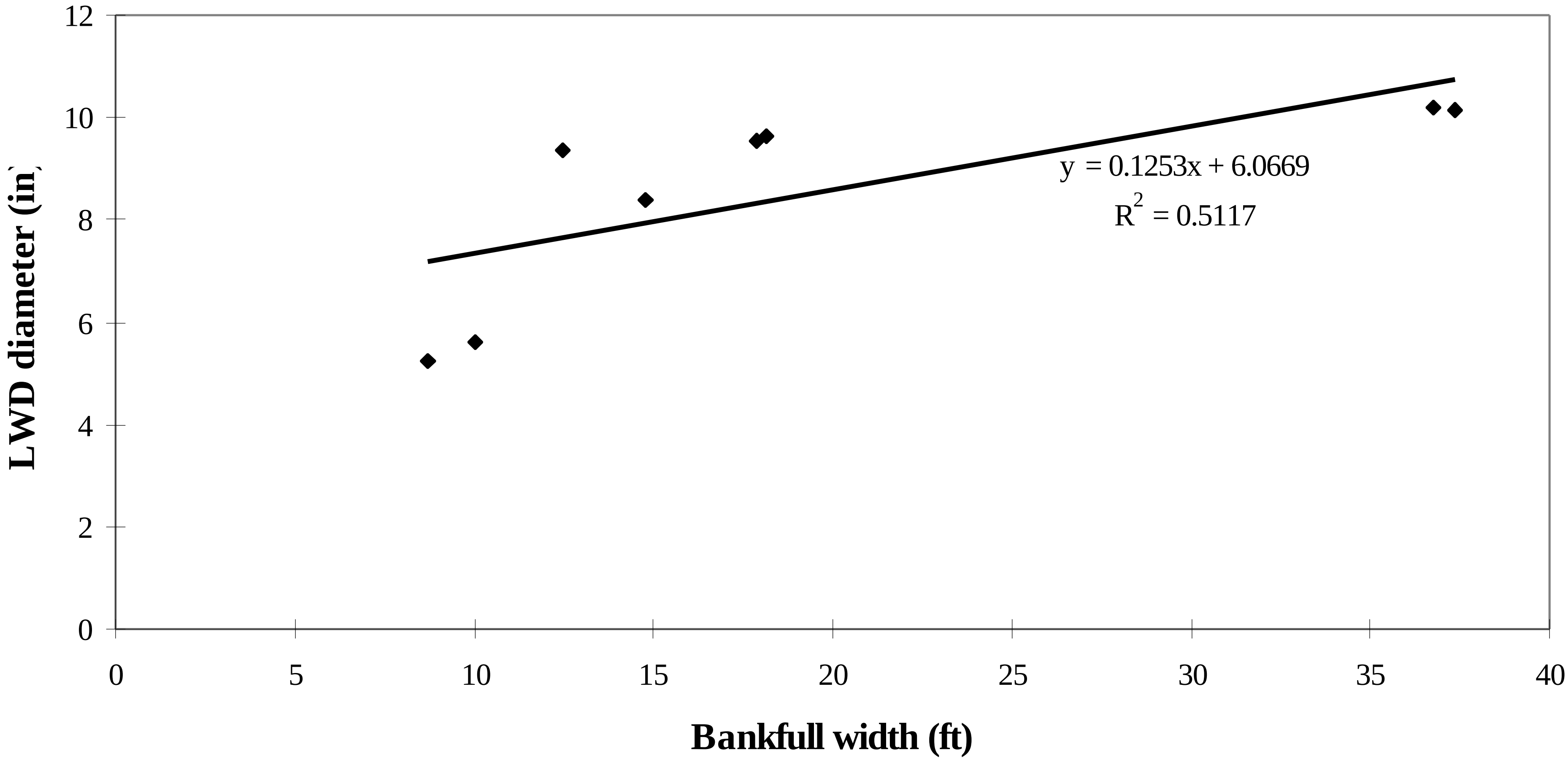


Figure B-11. Oregon – Knight. Unmanaged only.

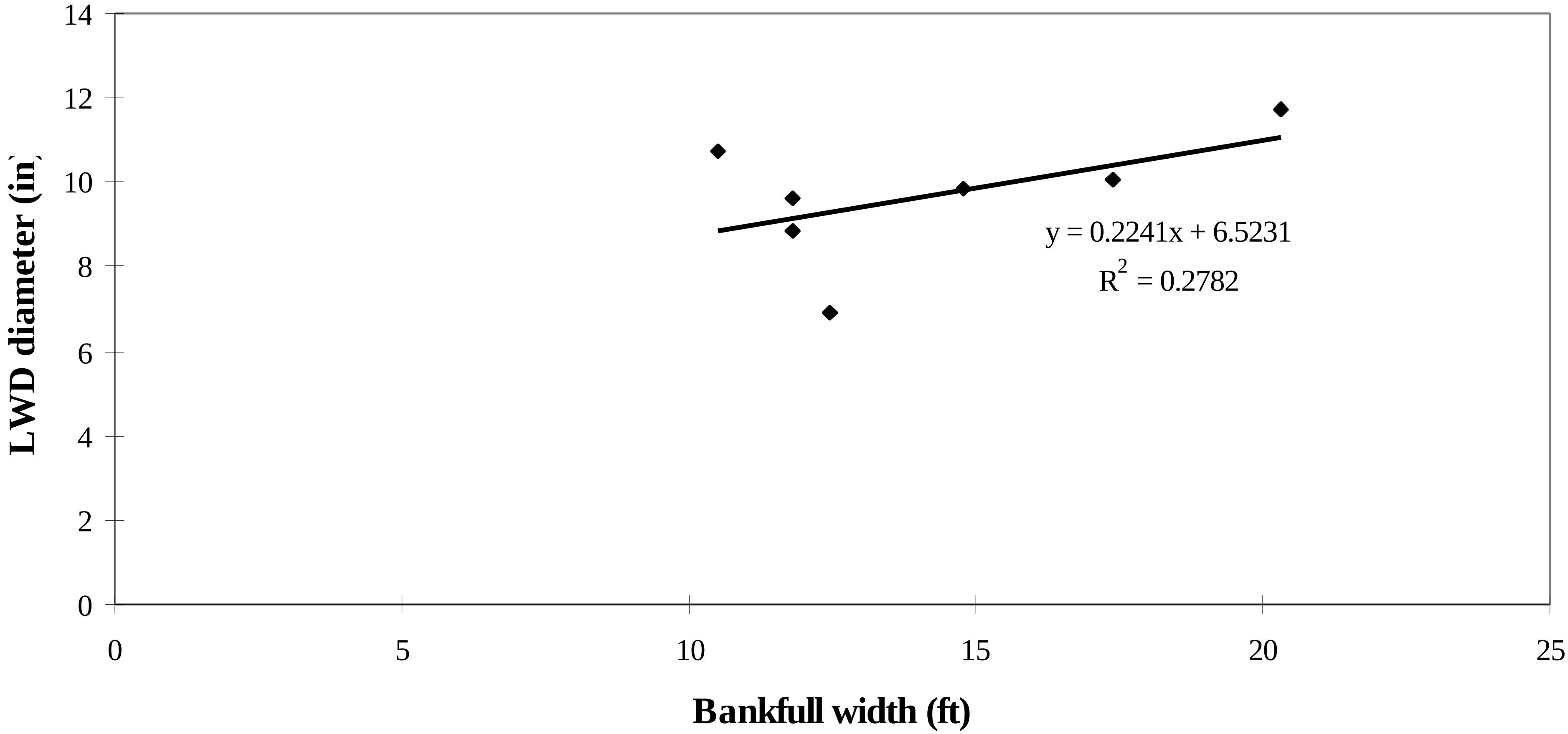


Figure B-12. Swan & Thompson Data. Undisturbed only.

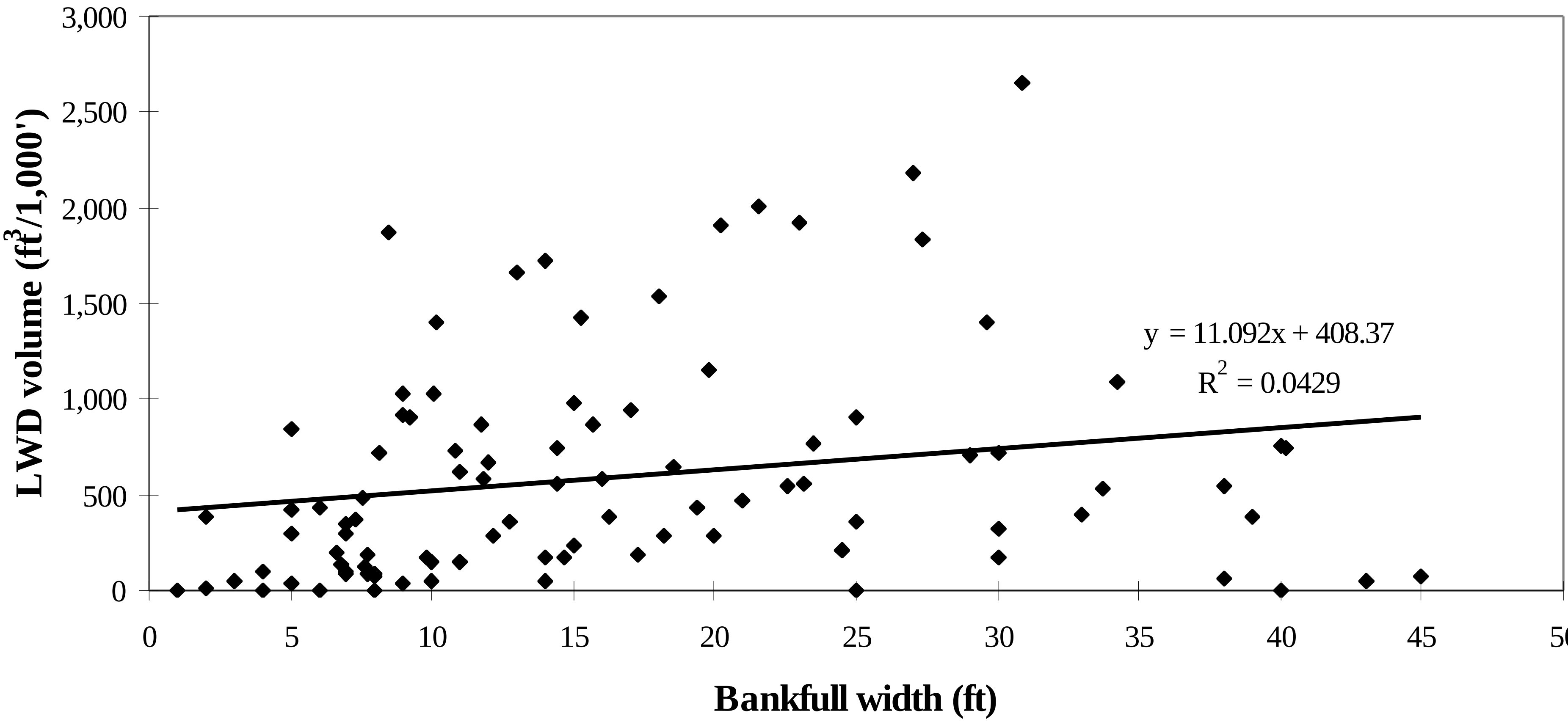


Figure B-13. Bilby — E. Washington. Unmanaged only.

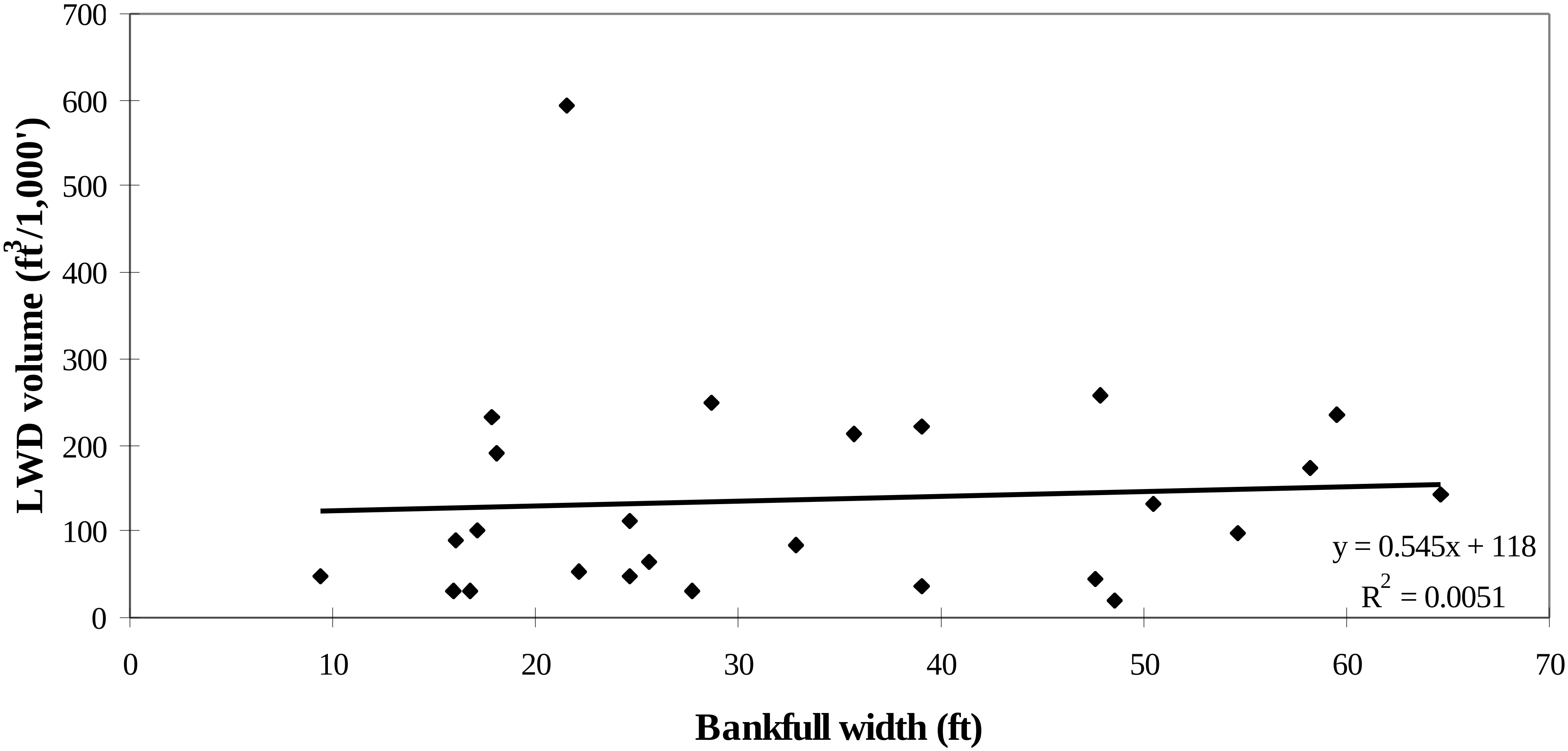


Figure B-14. Montana – Hayes.

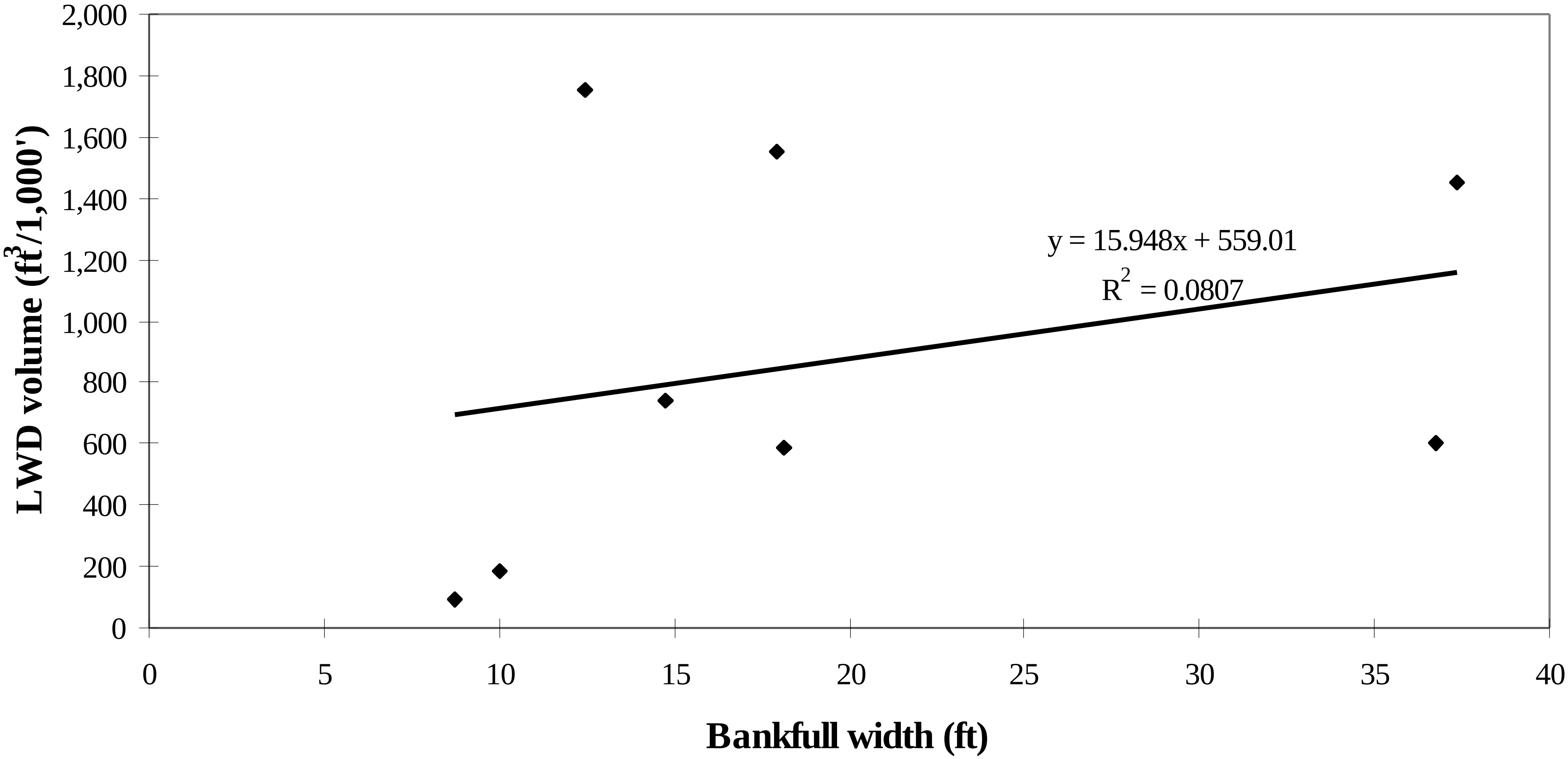


Figure B-15. Oregon – Knight. Unmanaged only.

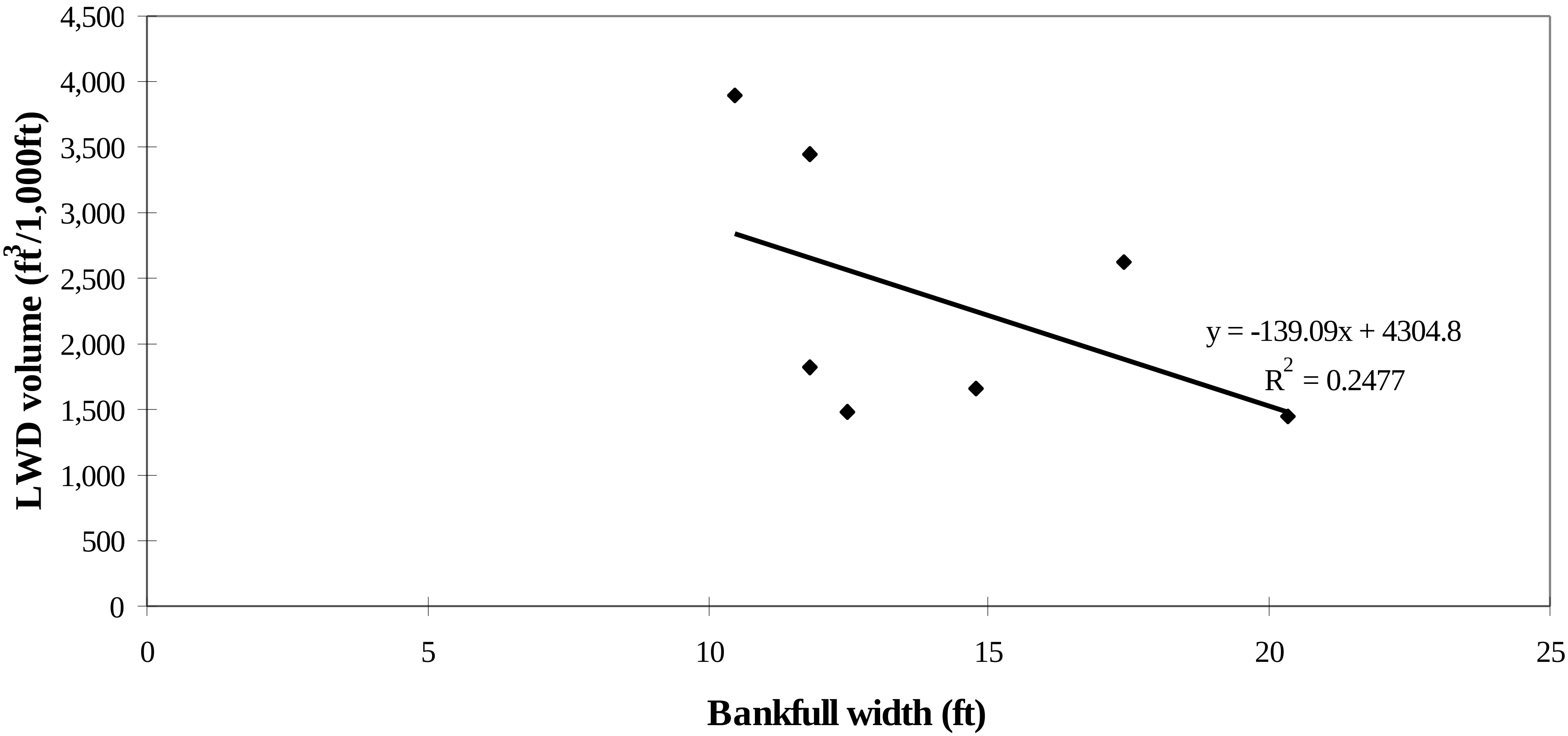


Figure B-16. Swan & Thompson Data. Undisturbed only.

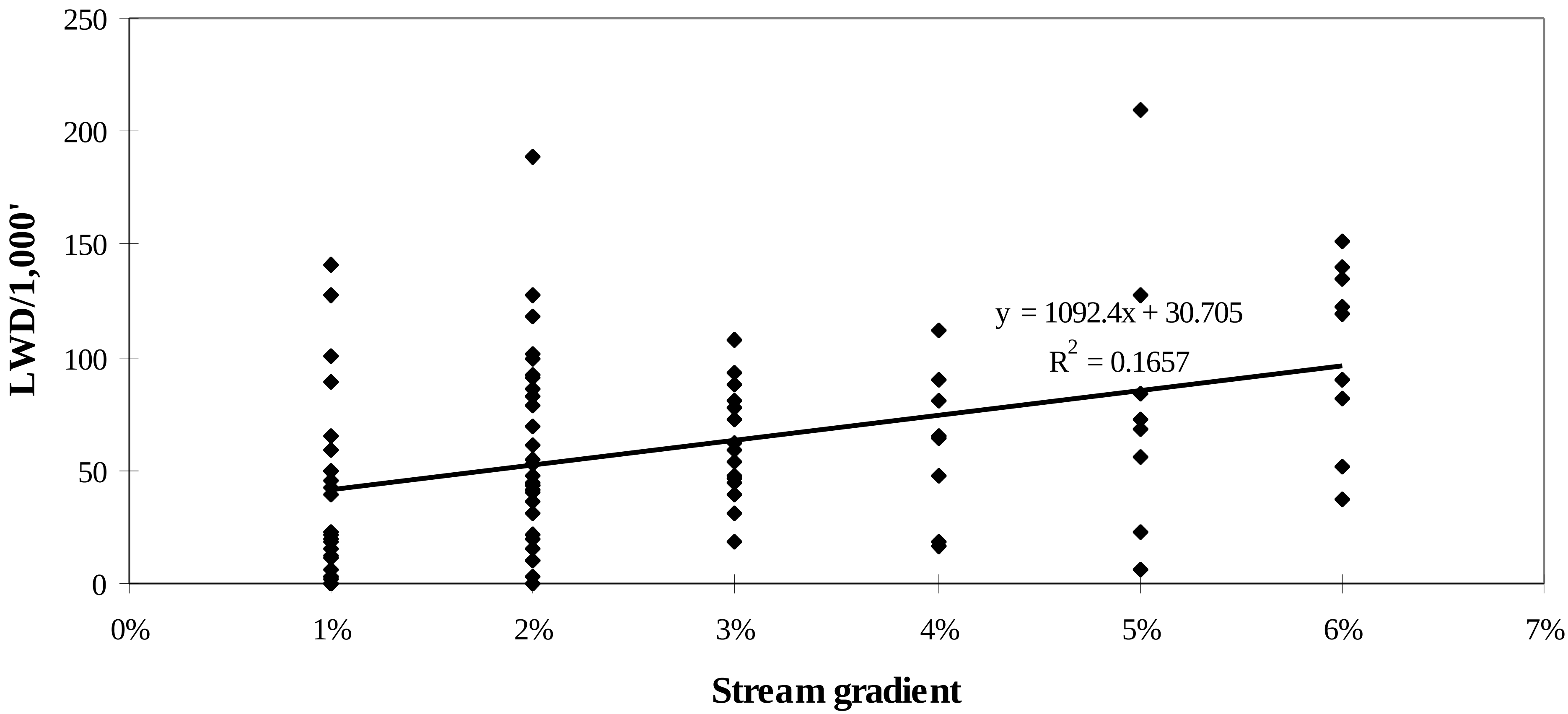


Figure B-17. Colorado Unmanaged Data.

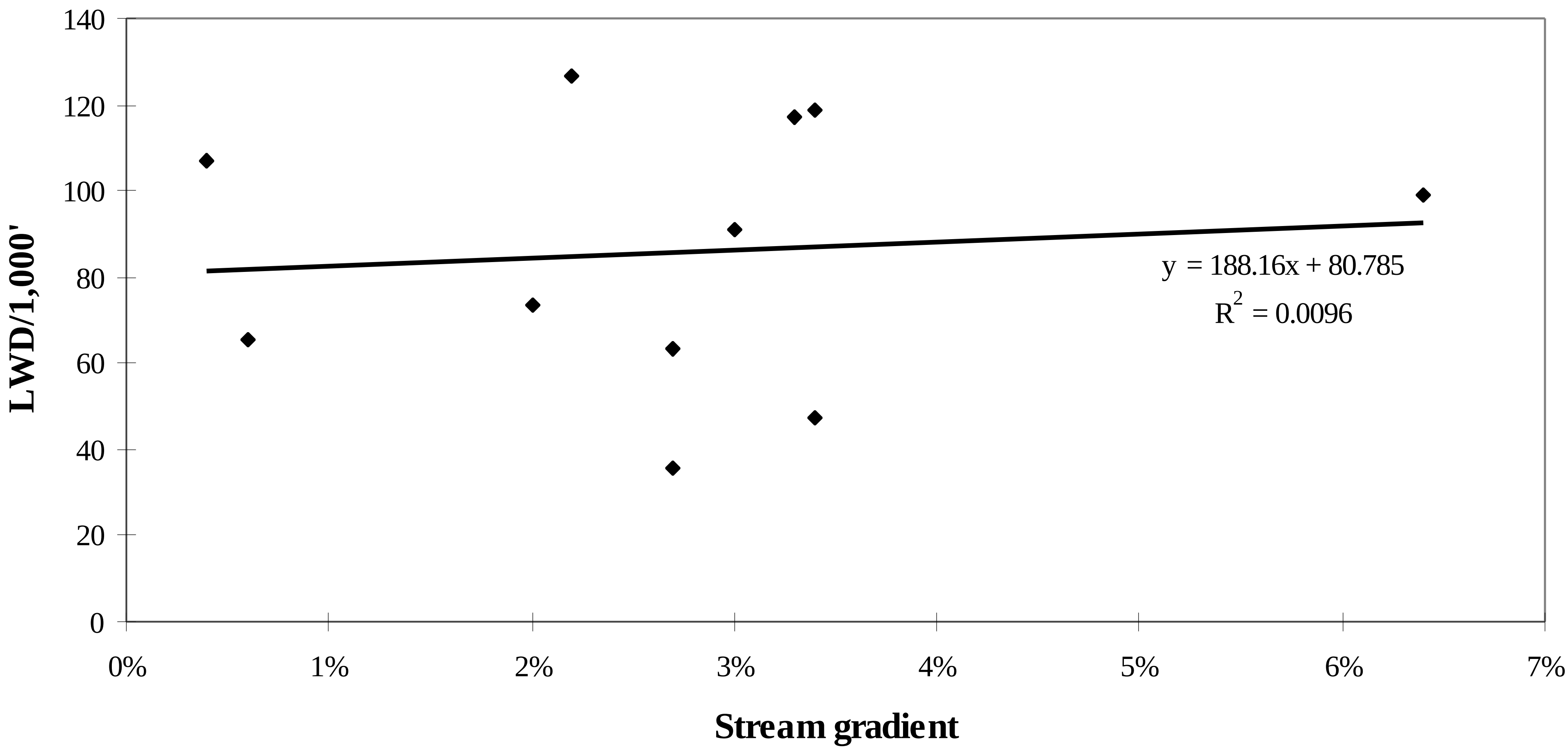


Figure B-18. Bitterroot National Forest.

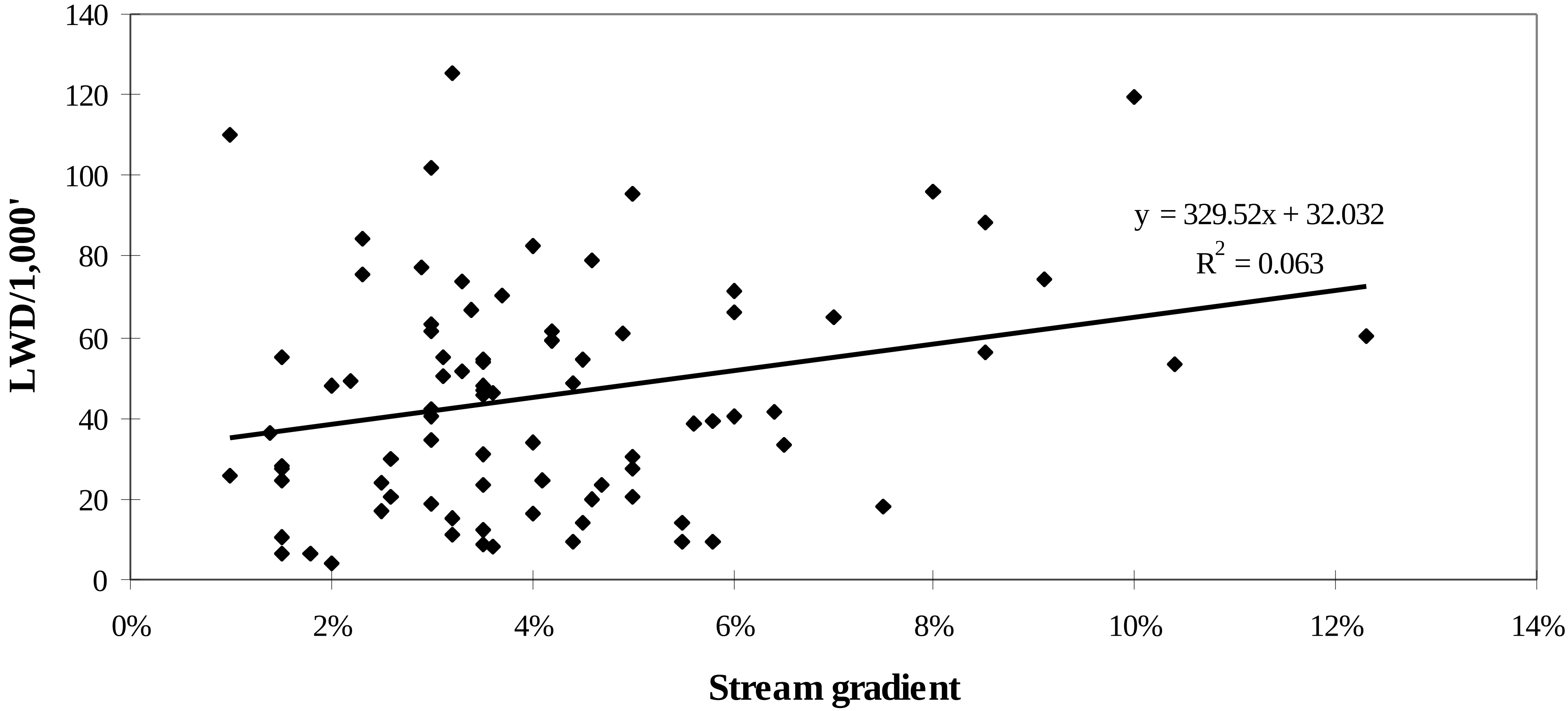


Figure B-19. Bilby — E. Washington. Unmanaged only.

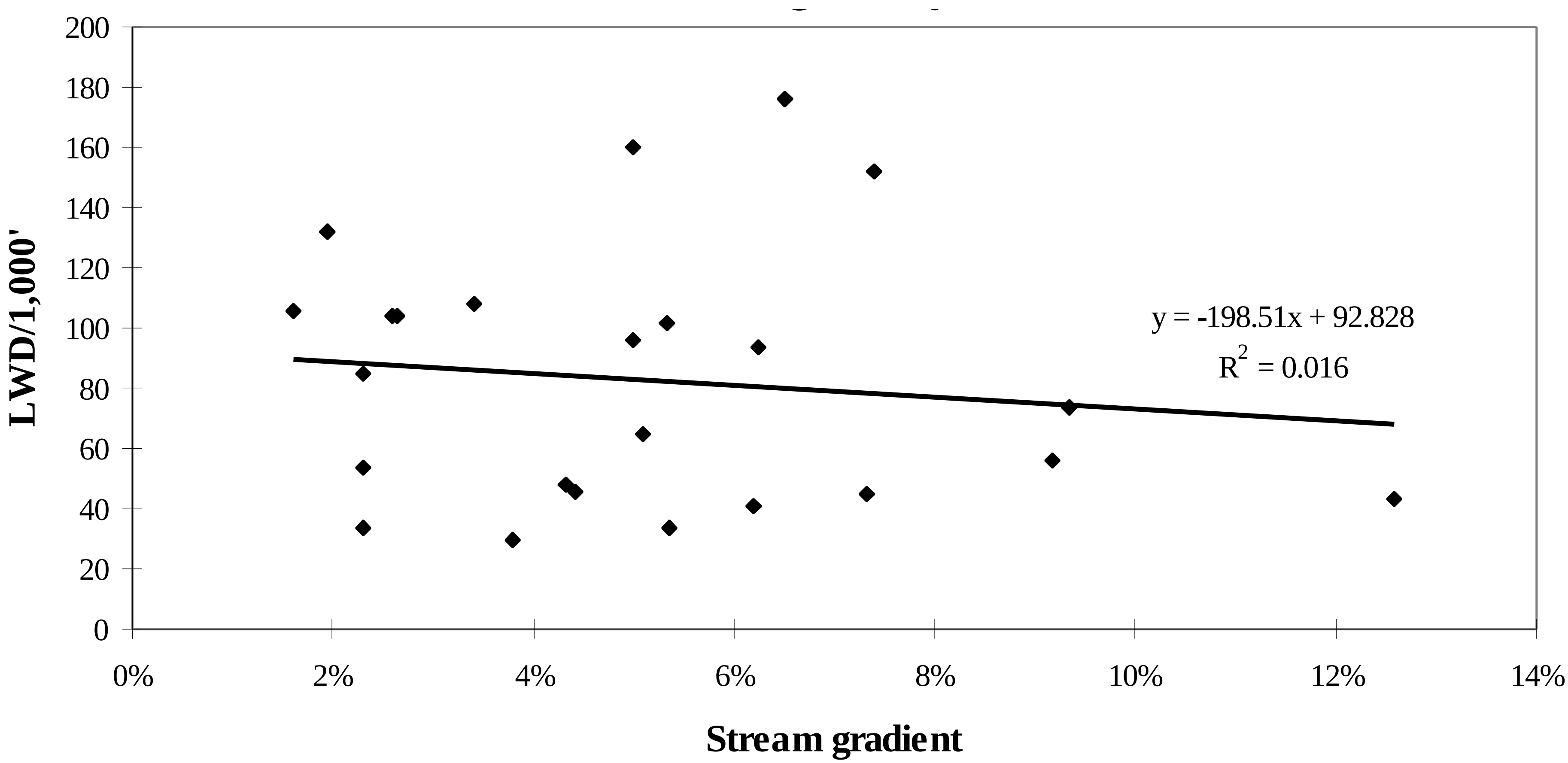


Figure B-20. Montana – Hayes.

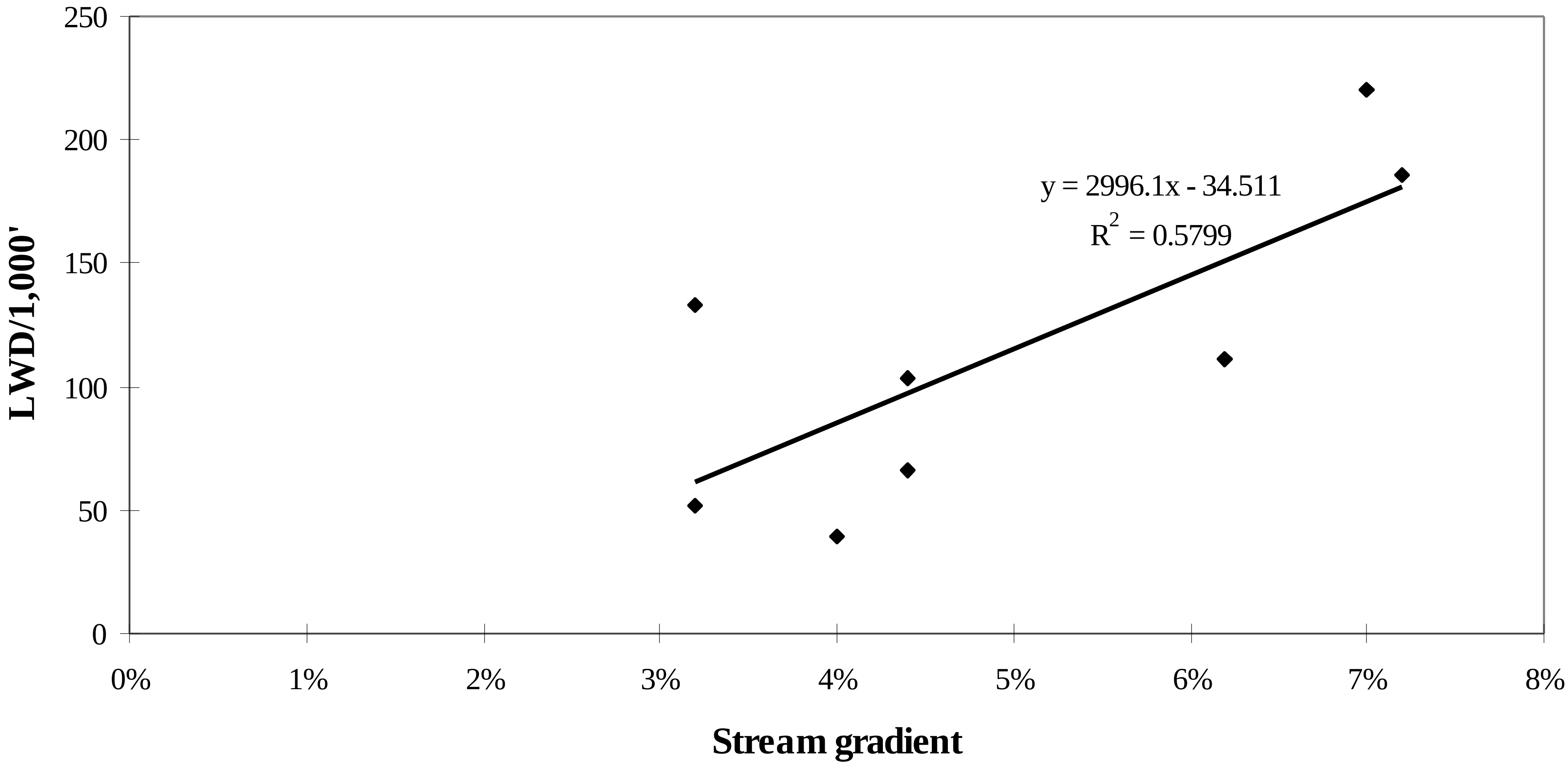


Figure B-21. Idaho (Overton).

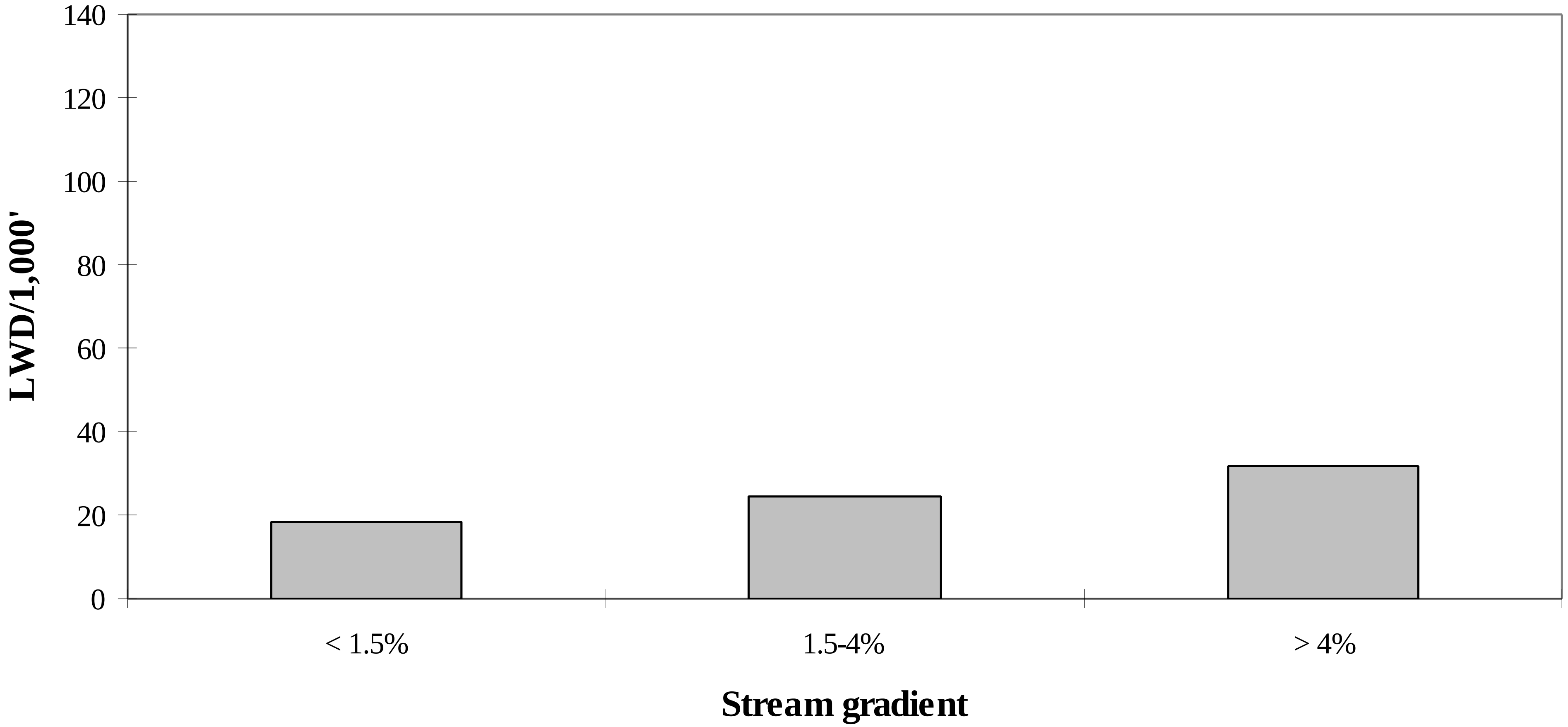


Figure B-22. Huntington Data. Unmanaged only.

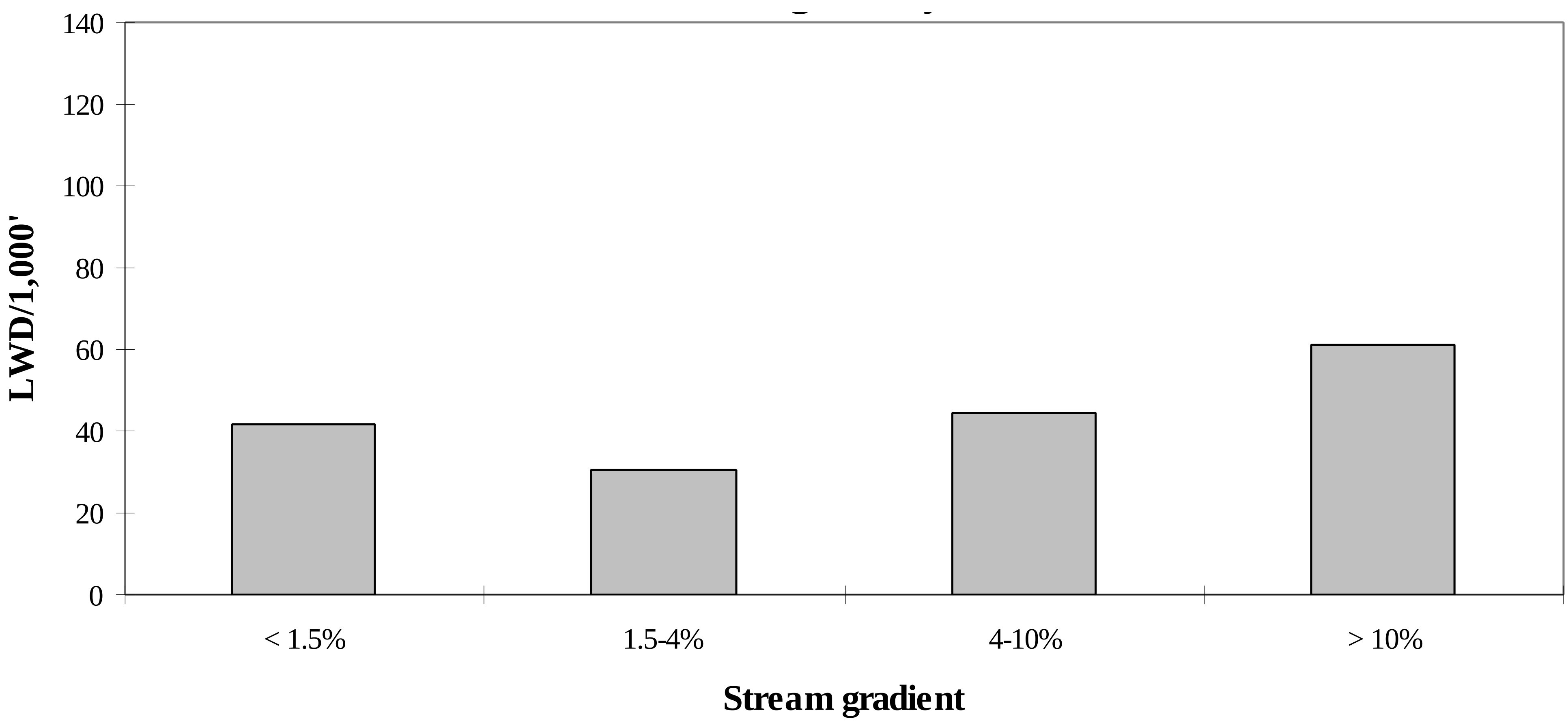


Figure B-23. Swan & Thompson Data. Undisturbed only.

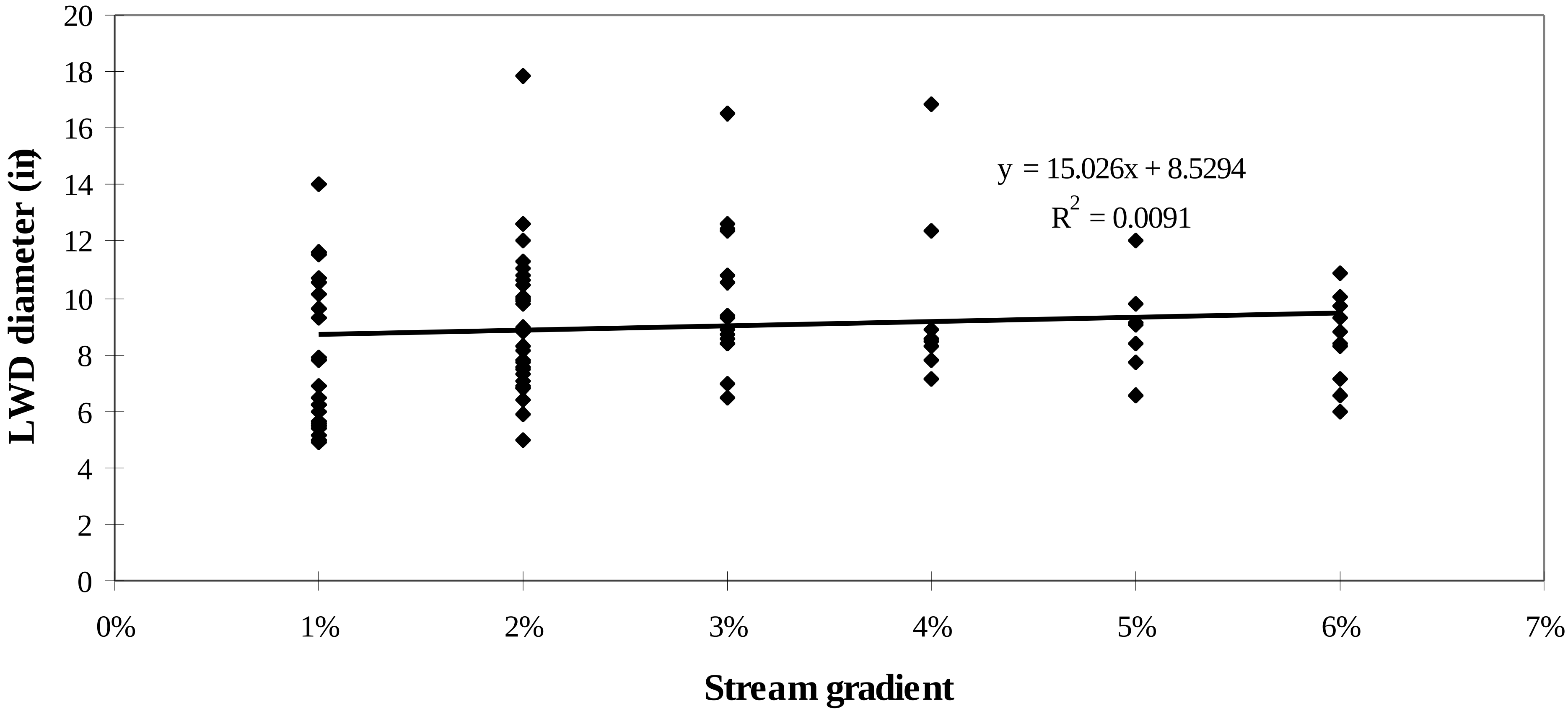


Figure B-24. Bilby — E. Washington. Unmanaged only.

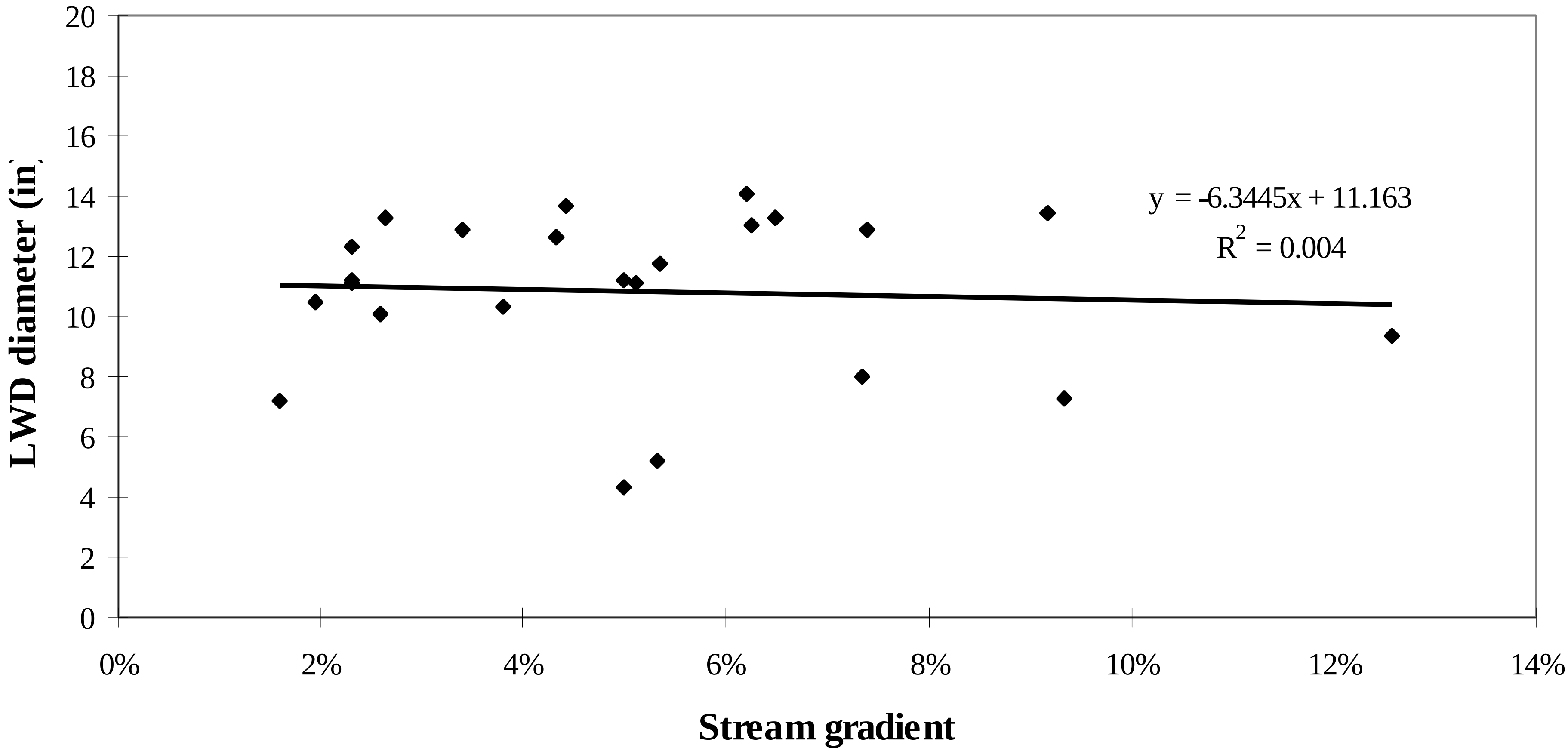


Figure B-25. Montana – Hayes.

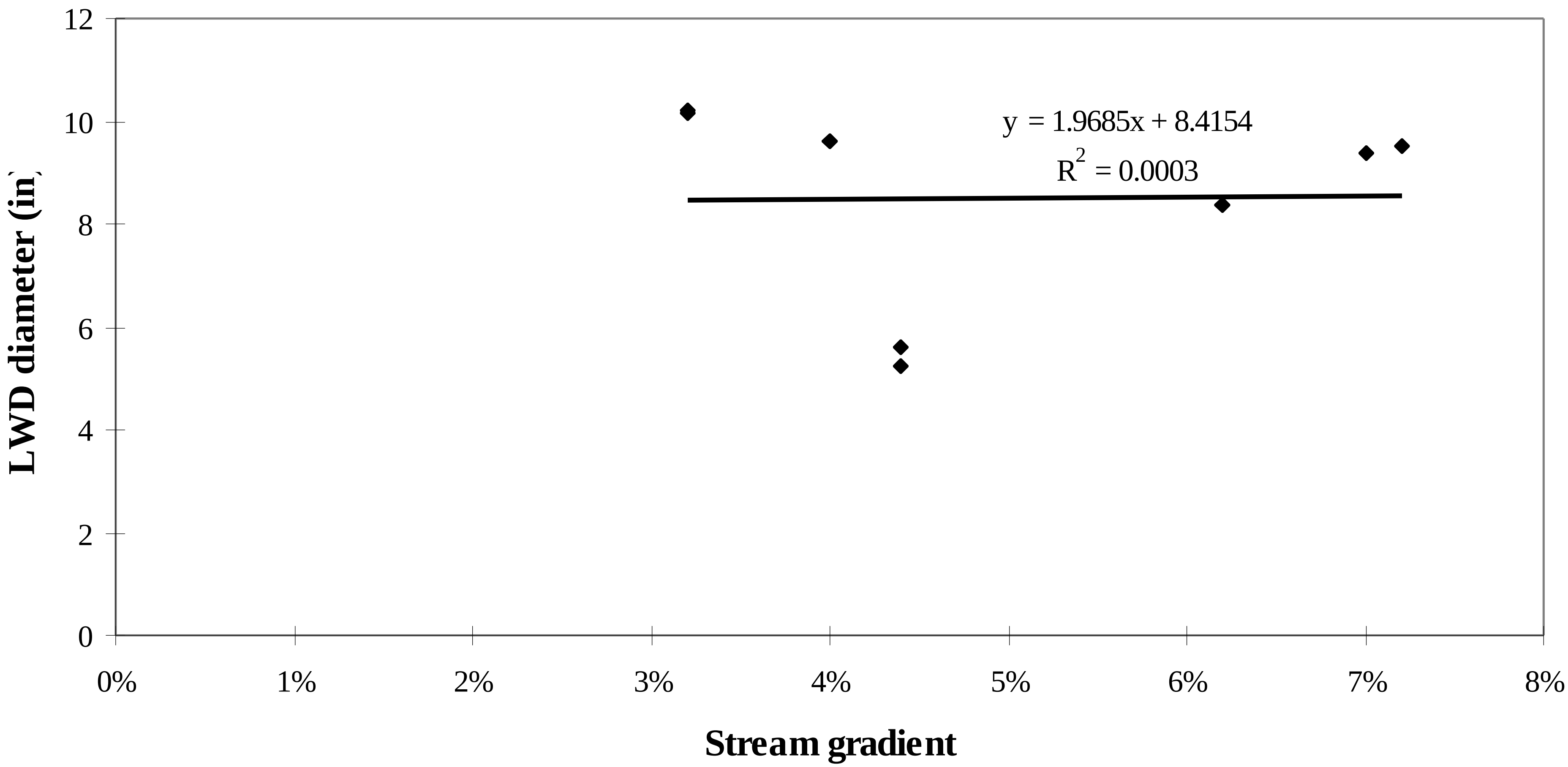


Figure B-26. Oregon – Knight. Unmanaged only.

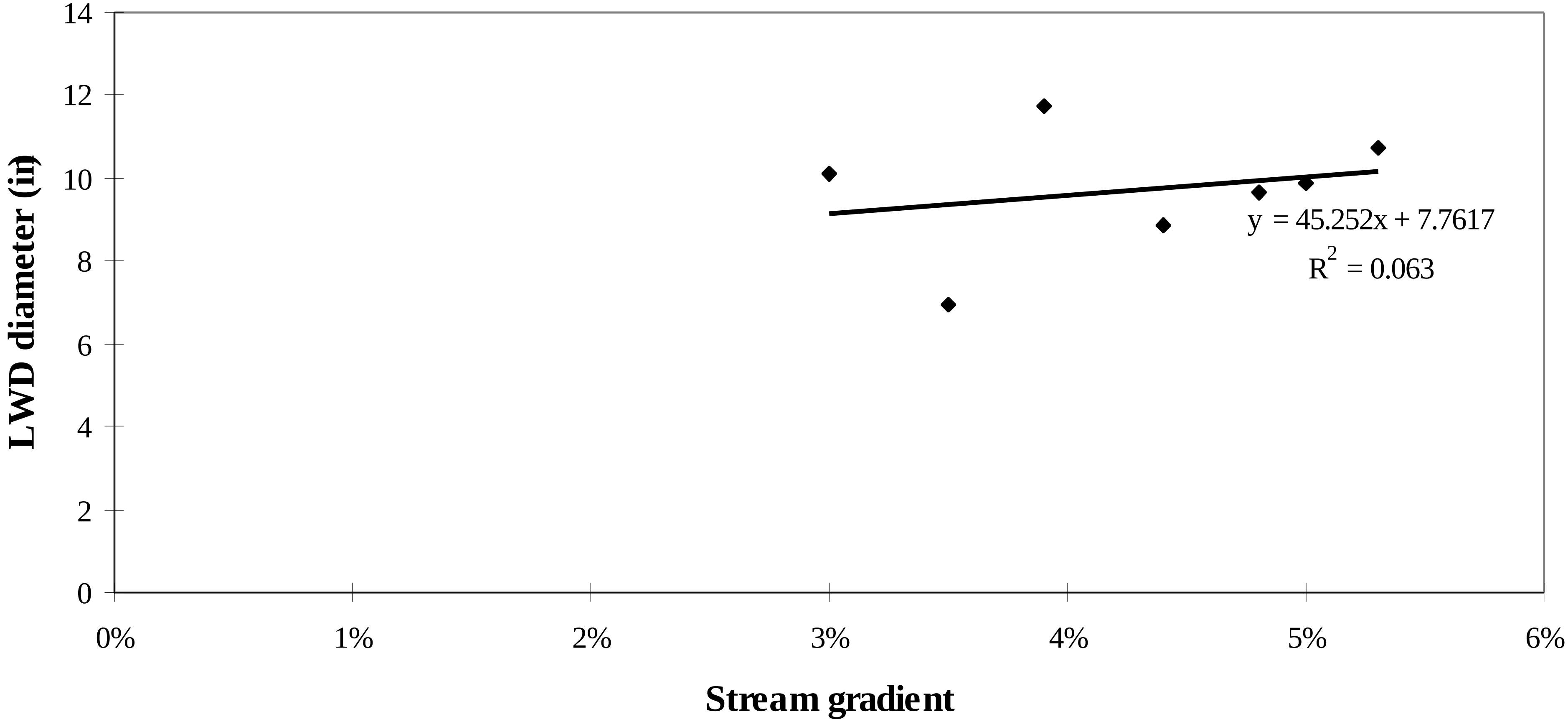


Figure B-27. Swan & Thompson Data. Undisturbed only.

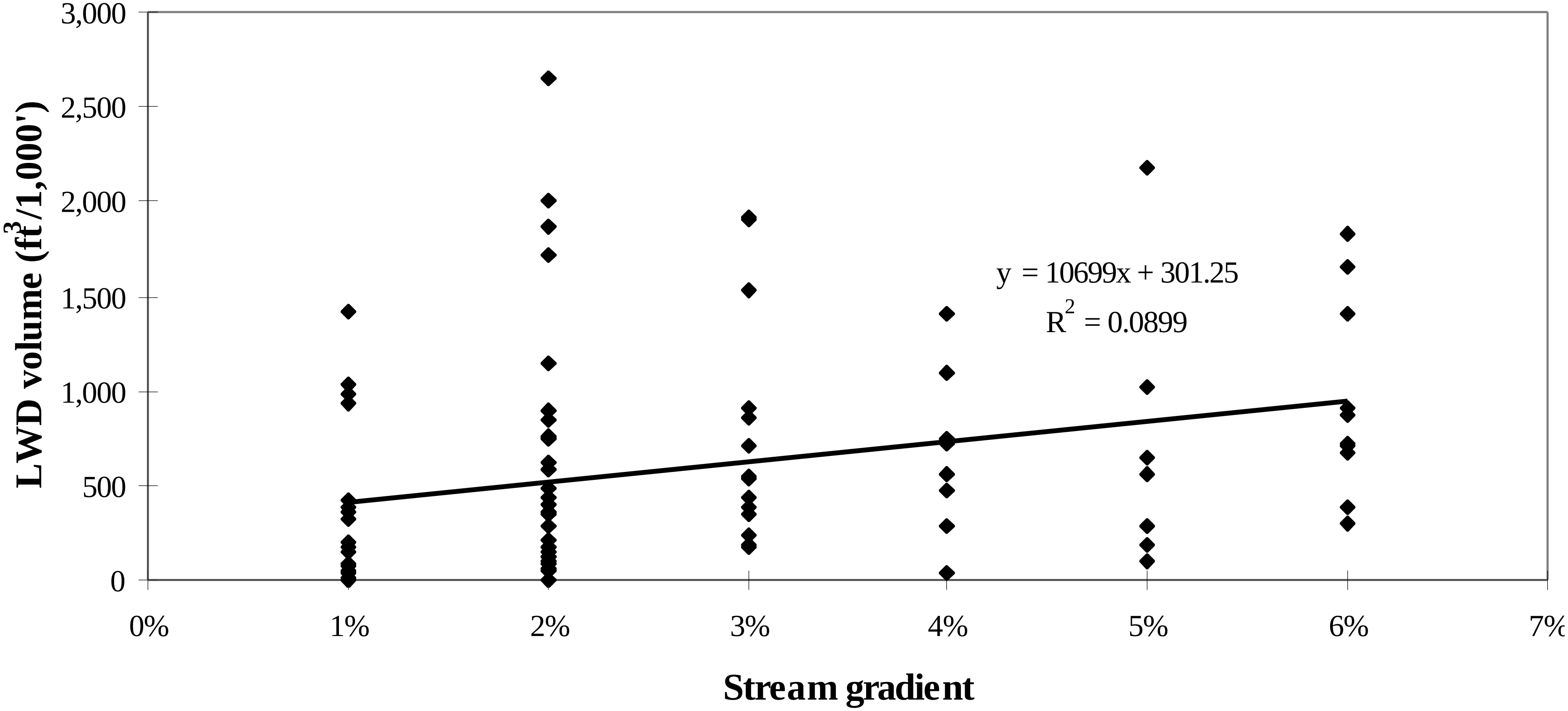


Figure B-28. Bilby — E. Washington. Unmanaged only.

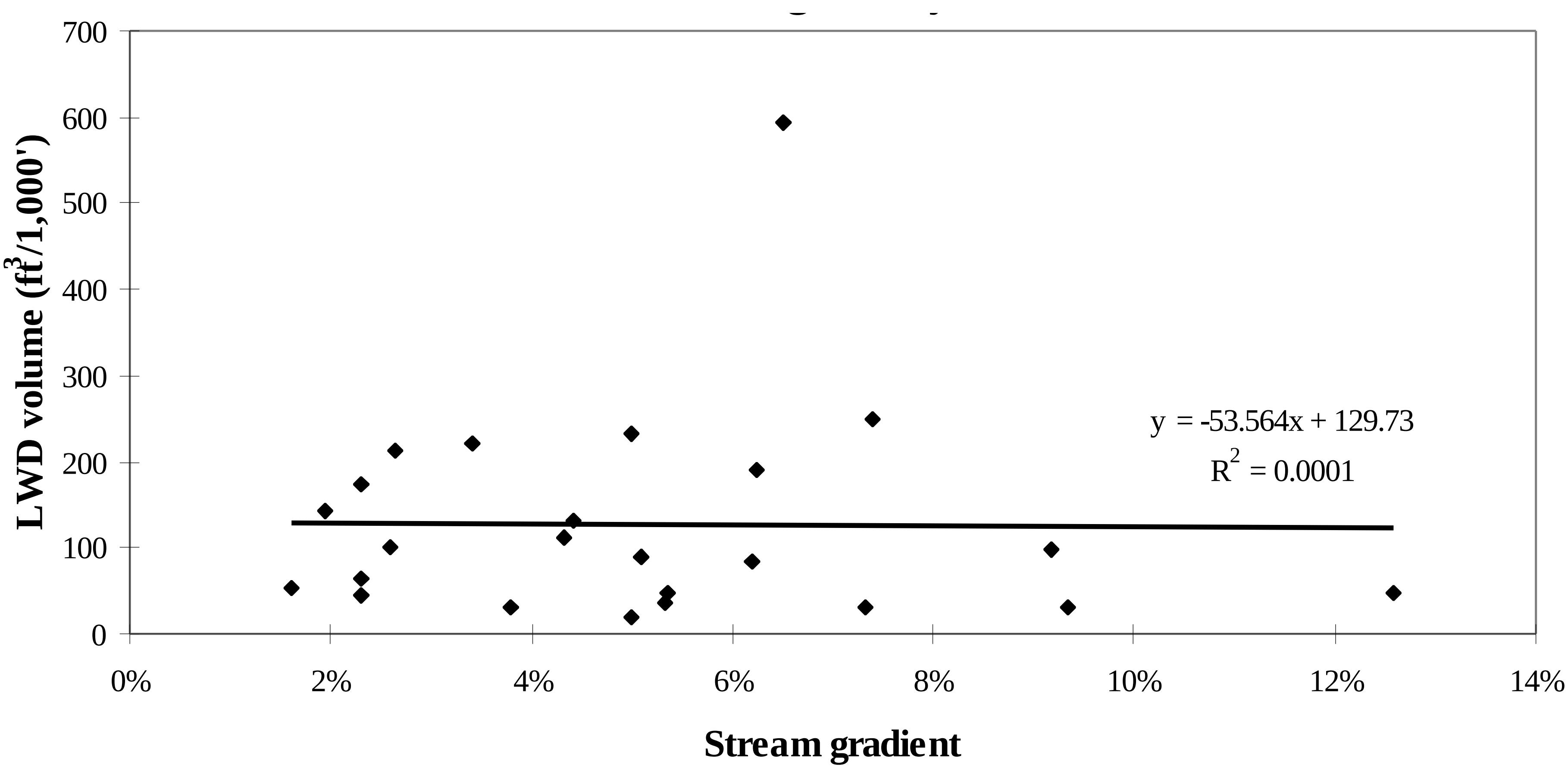


Figure B-29. Montana – Hayes.

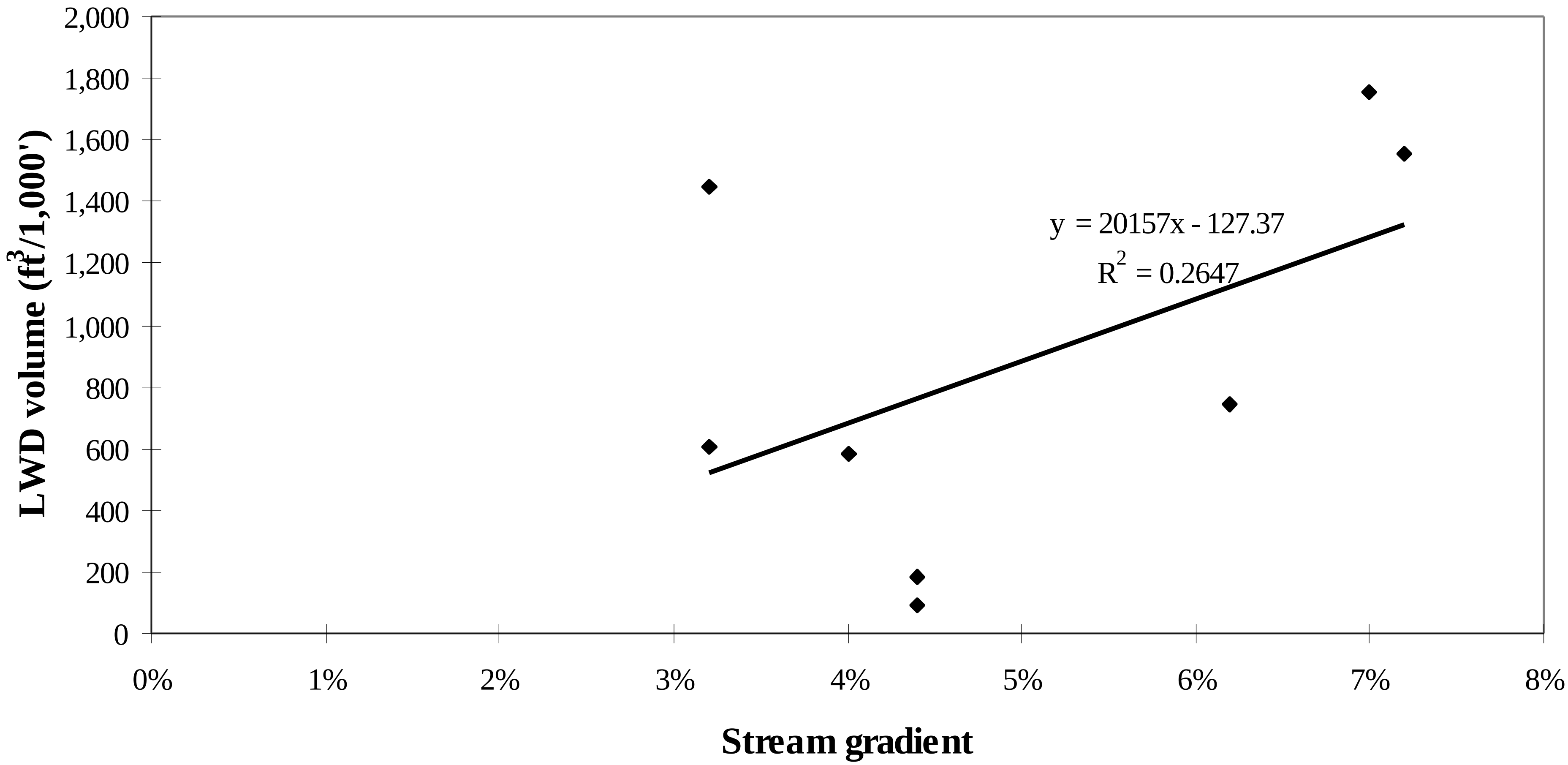


Figure B-30. Oregon – Knight. Unmanaged only.

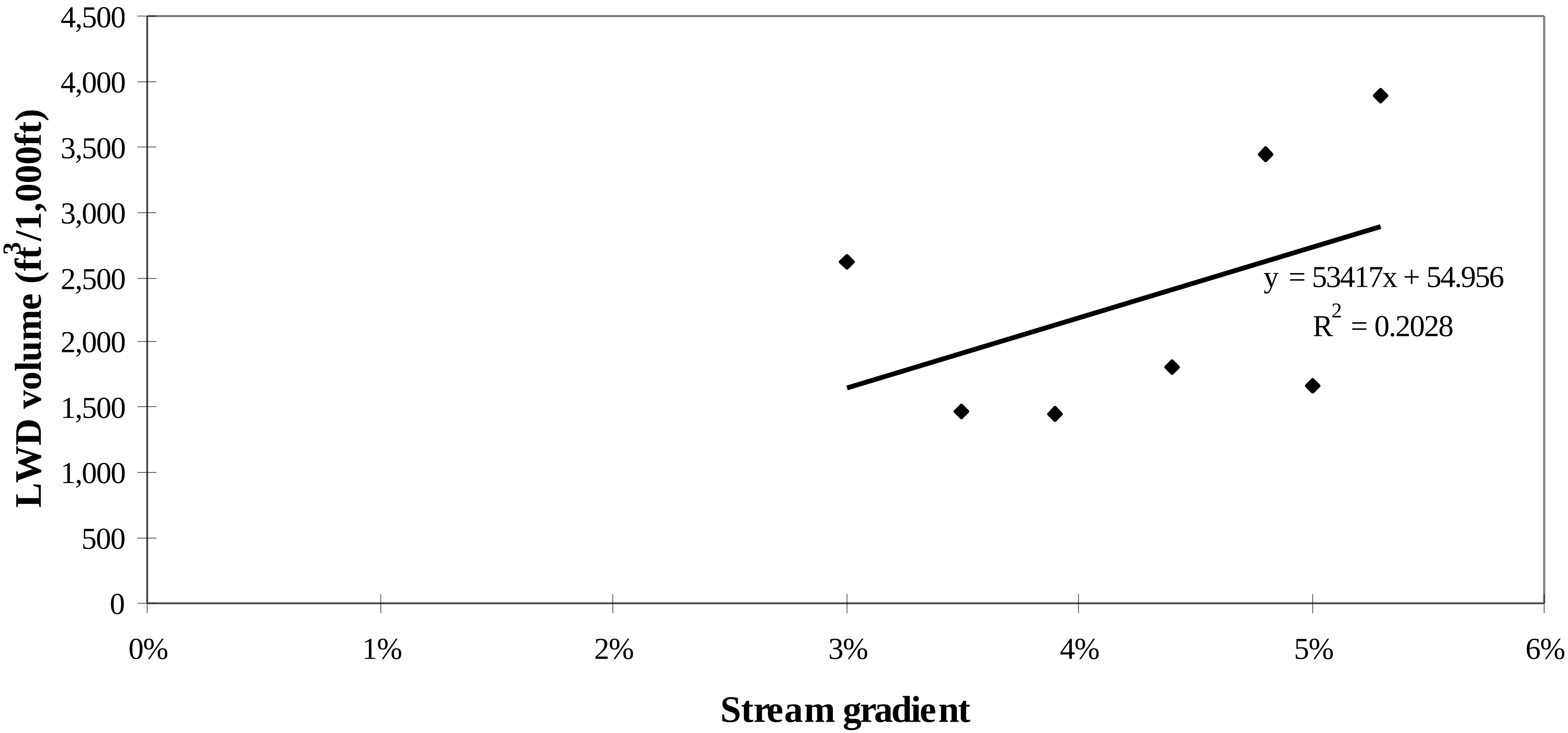


Figure B-31. Swan & Thompson Data. Undisturbed only.

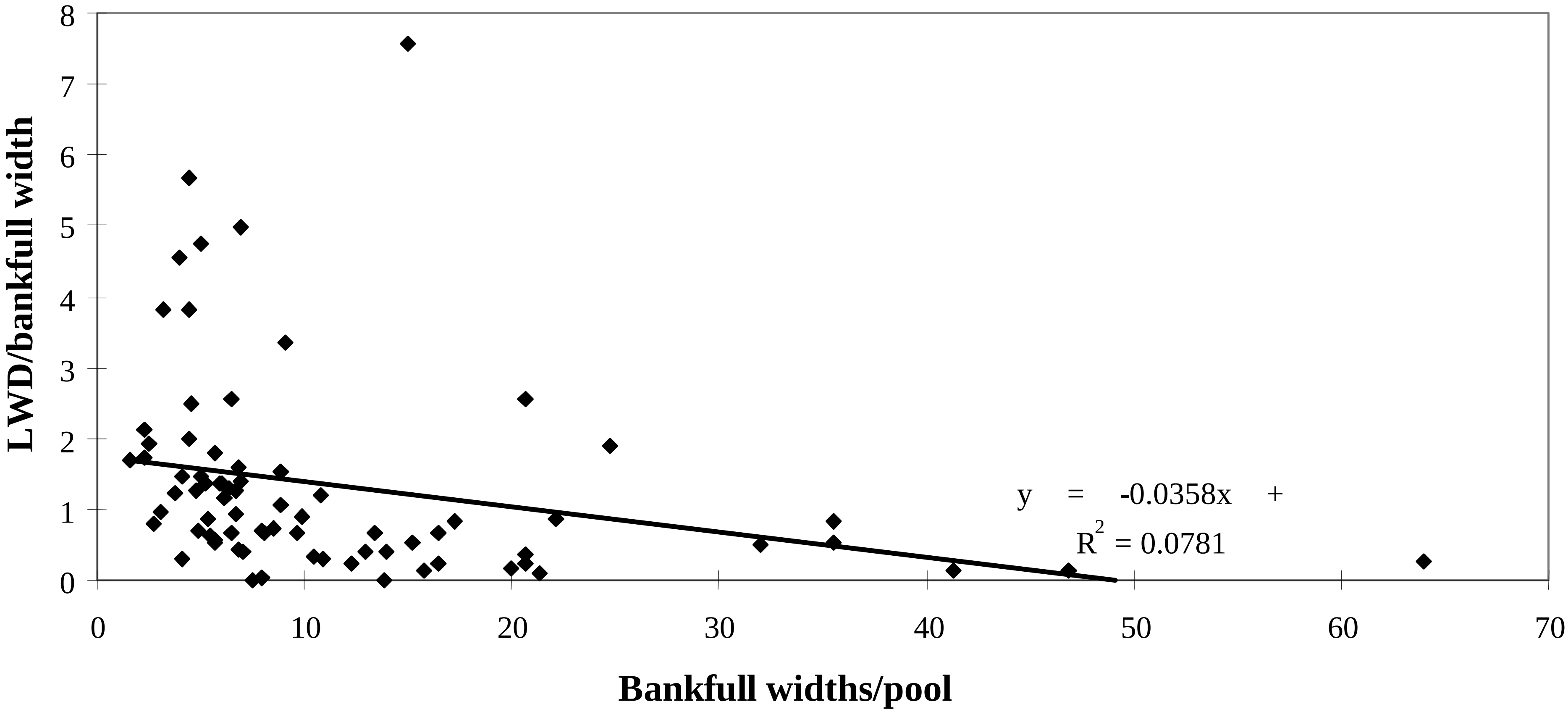


Figure B-32. Colorado Unmanaged Data.

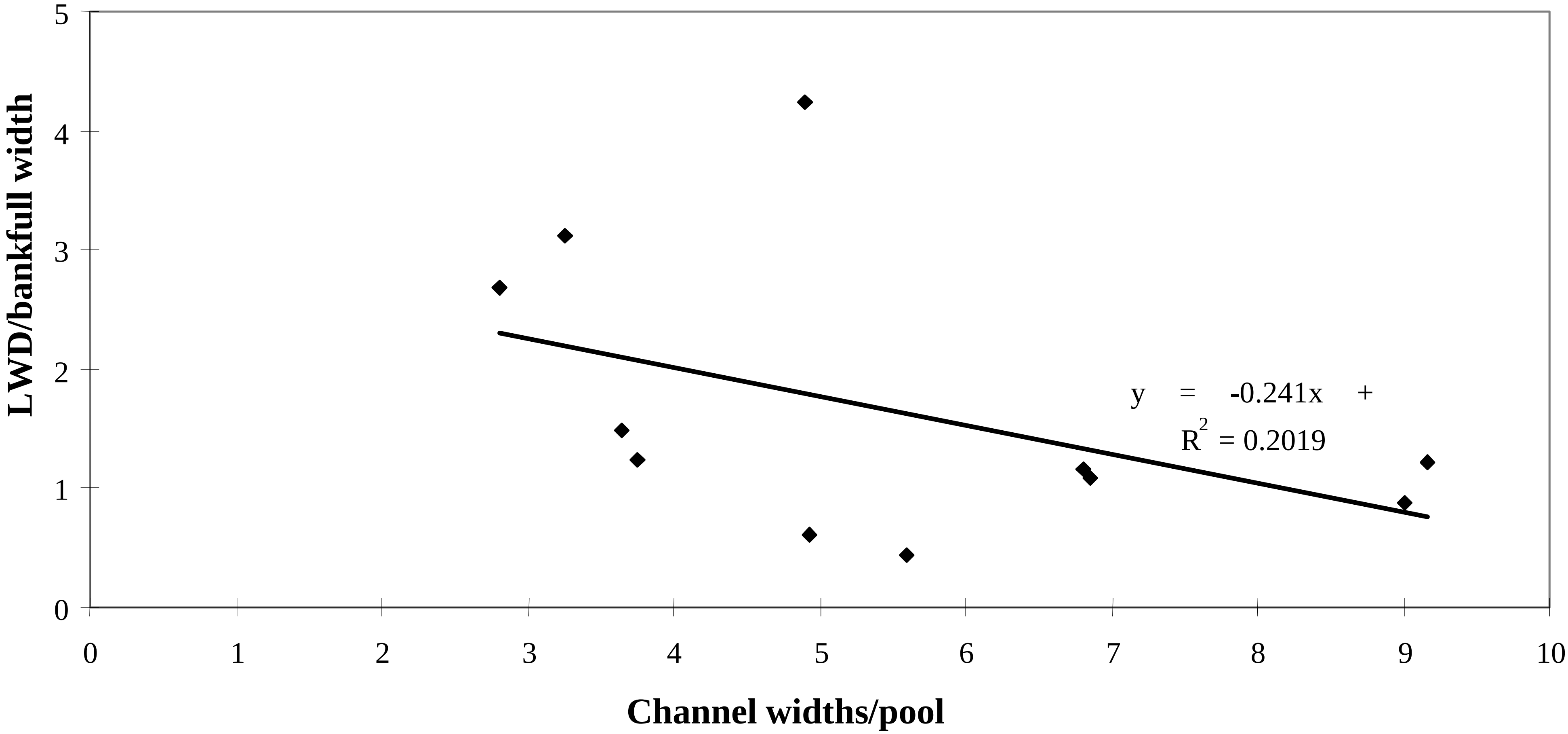


Figure B-33. Bitterroot National Forest.

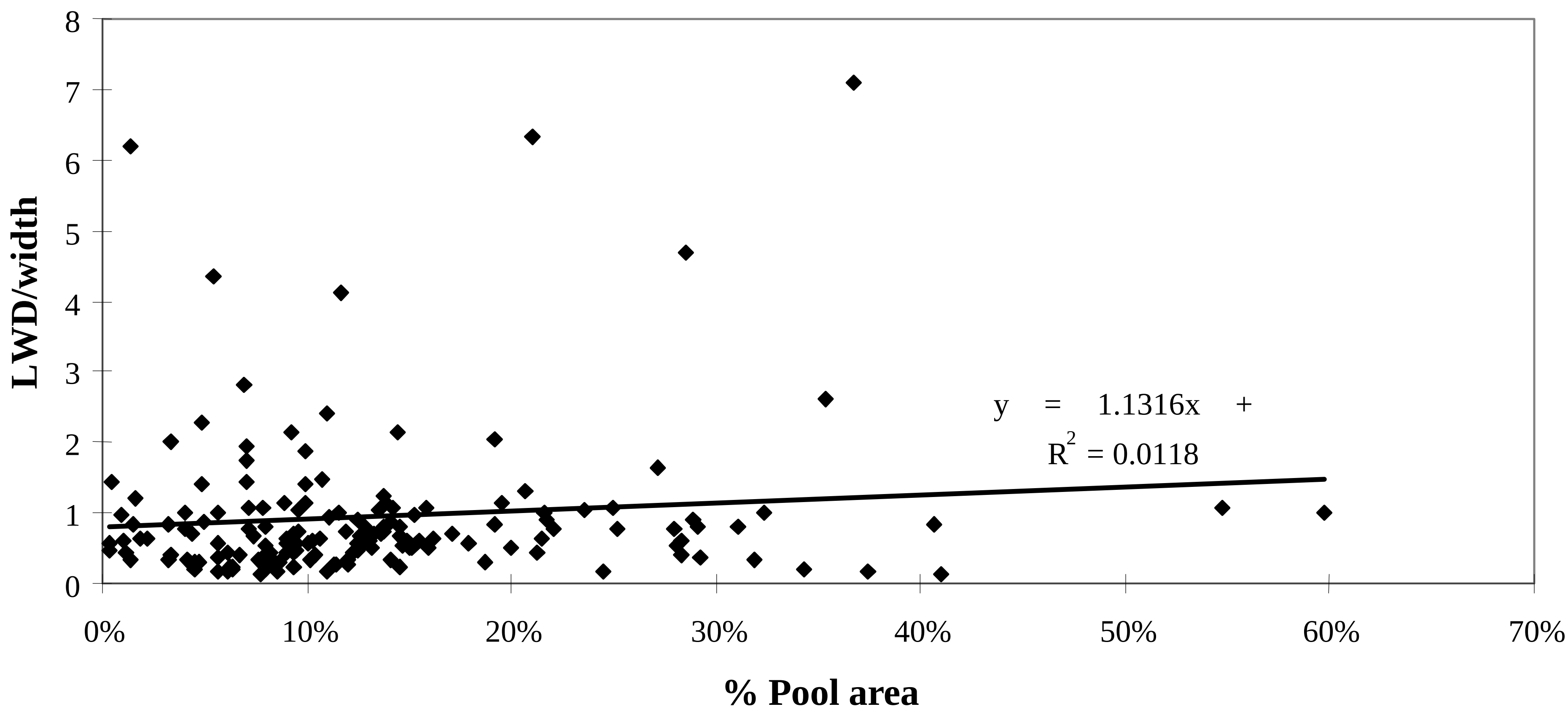


Figure B-34. Idaho (Overton)

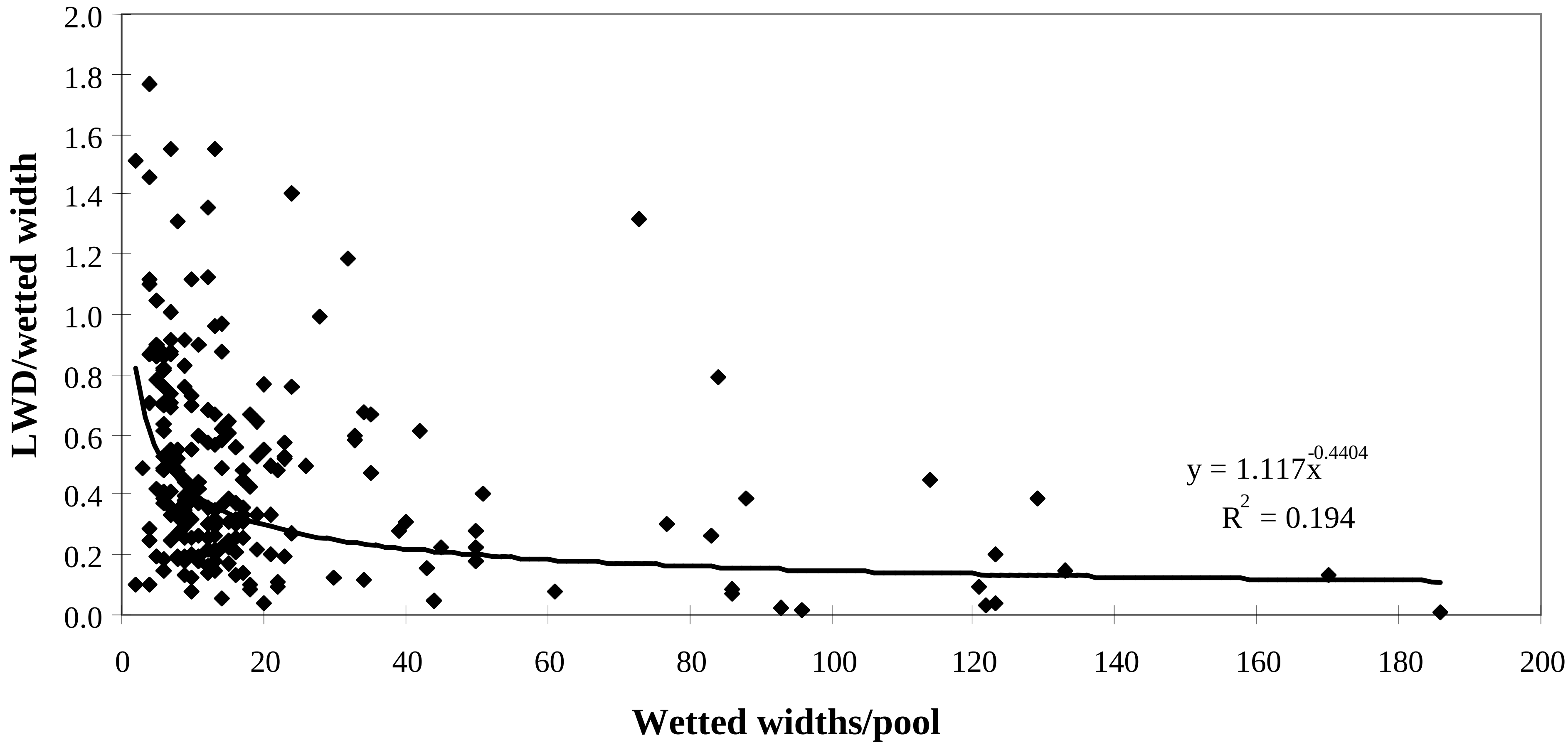


Figure B-35. Huntington Data.

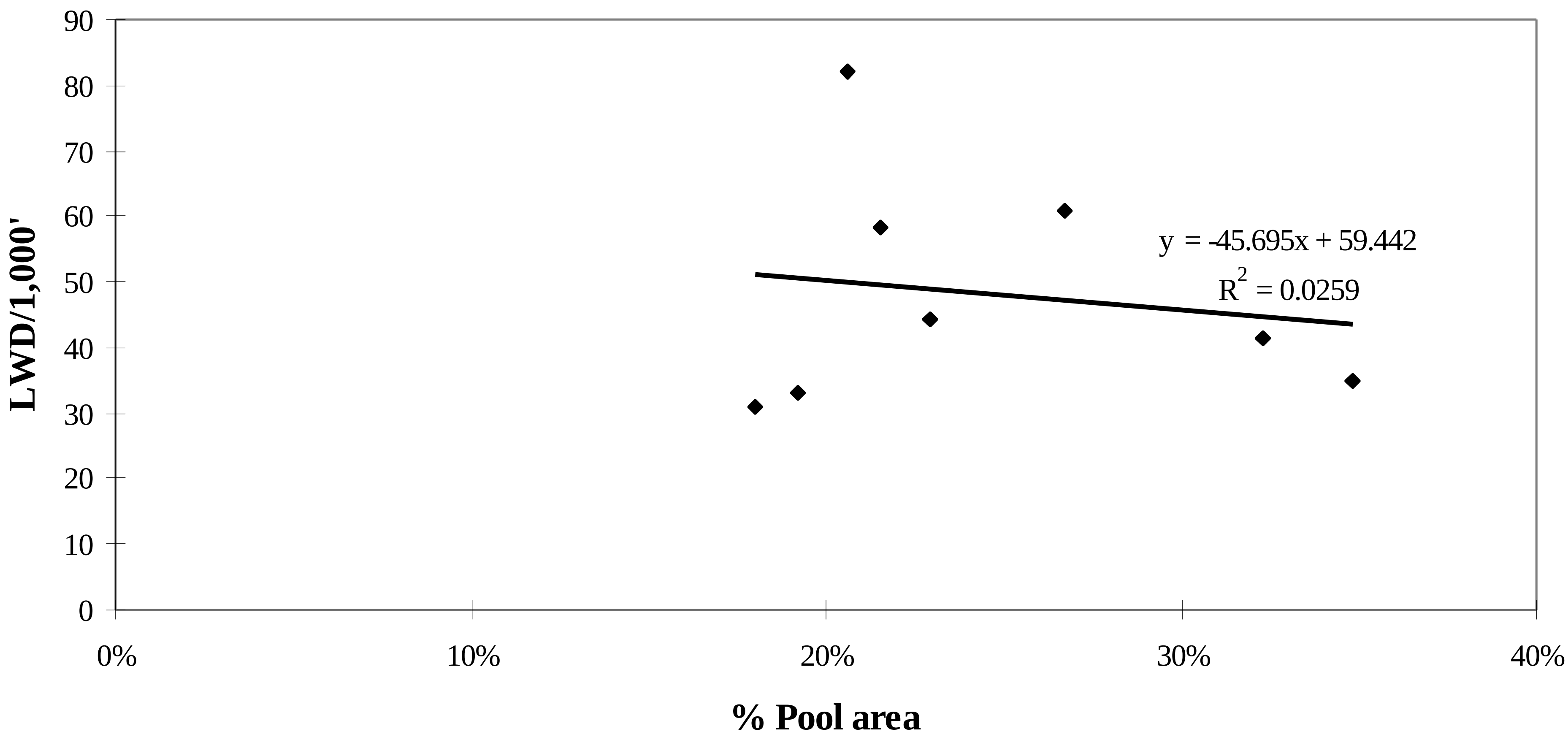


Figure B-36. Oregon – Knight. Unmanaged only.

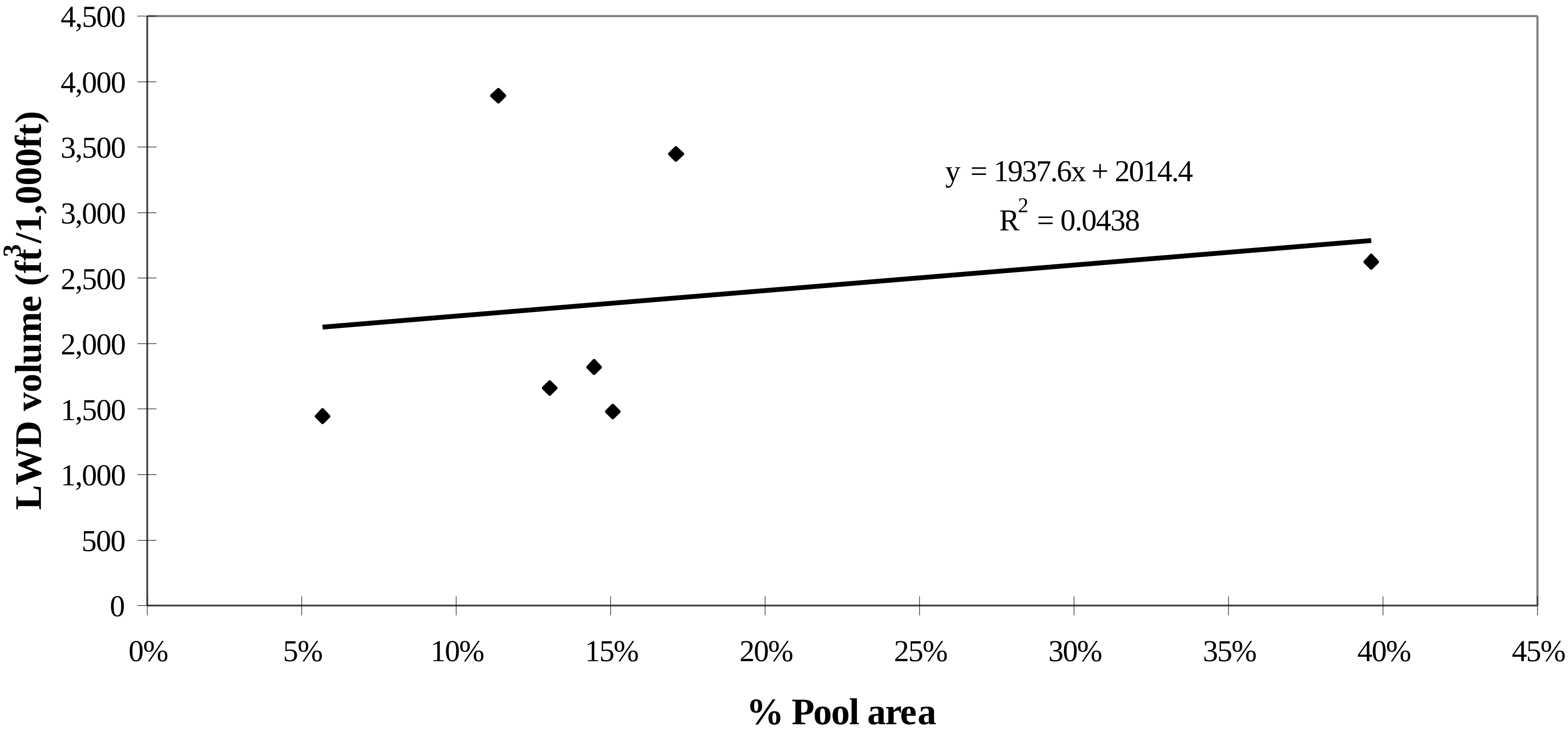


Figure B-37. LWD step frequency as a function of channel gradient in western North America.

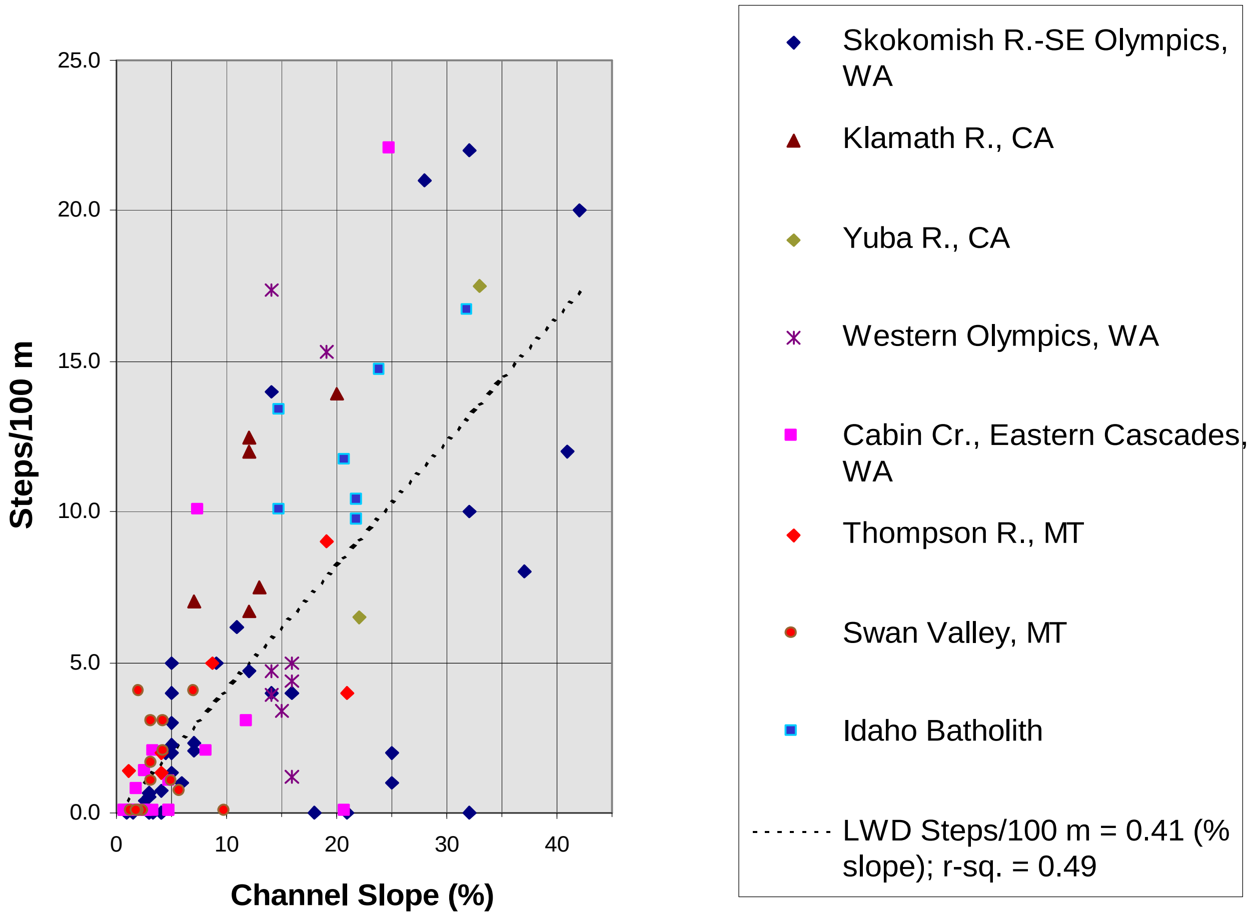


Figure B-38. LWD step height as a function of channel gradient.

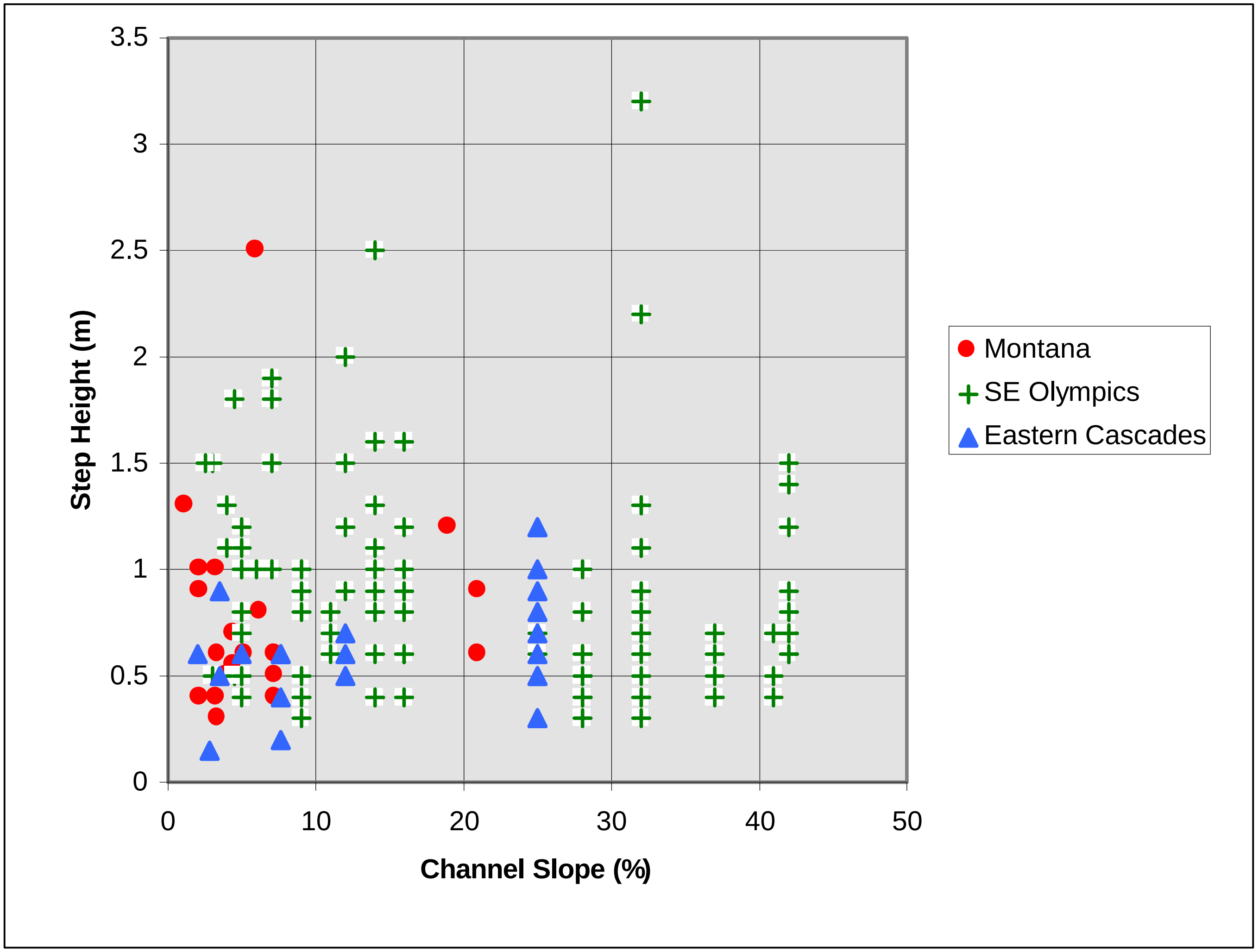
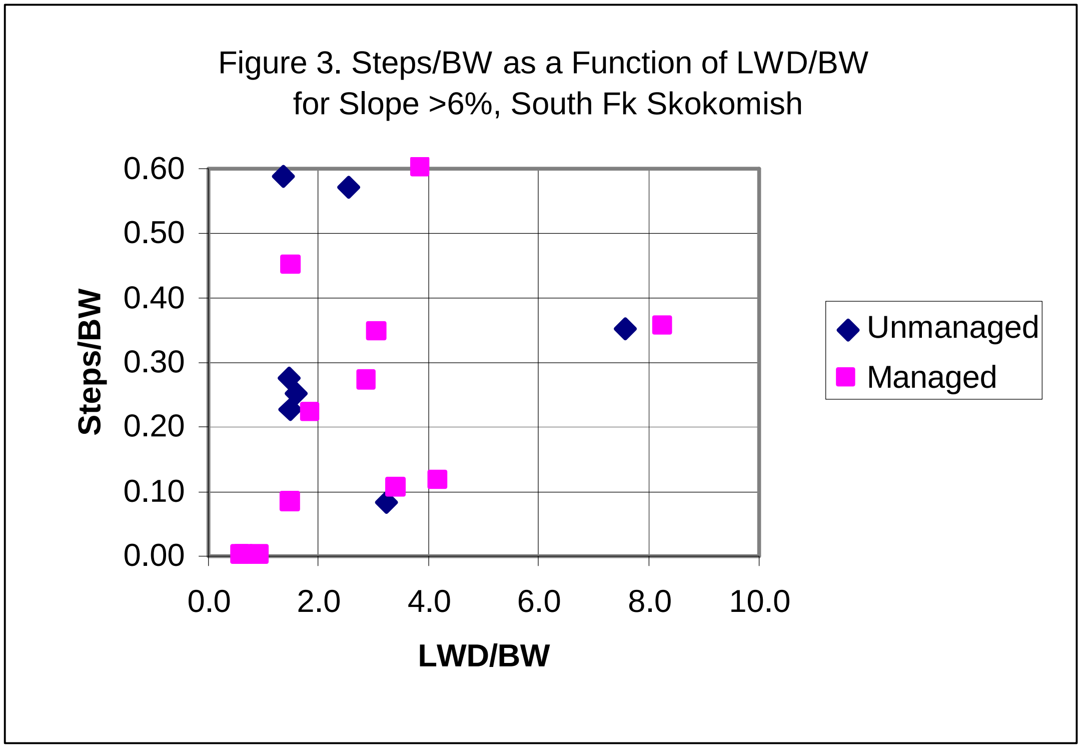


Figure B-39. Steps per bankfull width as a function of LWD pieces per bankfull width for streams with gradient >6%, South Fork Skokomish R., Washington



APPENDIX C

CHARACTERISTICS OF DOMINANT RIPARIAN STAND TYPES IN MONTANA, IDAHO, AND EASTERN WASHINGTON

March, 1999

TABLE OF CONTENTS

DOMINANT RIPARIAN STAND TYPES OF THE INTERIOR COLUMBIA RIVER BASIN

To obtain realistic estimates of large woody debris (LWD) inputs to streams under different riparian management scenarios, information on actual conditions of stands was gathered in the Interior Columbia River Basin portion of the NFHCP project area. Timber surveys were conducted in the following manner during winter 1998.

1.1 Sampling Design

Riparian sampling locations were drawn from the pool of perennial streams in Tier I watersheds[31] on Plum Creek Timber Company land east of the Cascade Mt. Crest. Sampling locations were chosen randomly by land sections (1 mi^2). Non-forested land identified in Plum Creek Timber Company's timber inventory database were excluded (54 stream miles). Also excluded were sections with less than 0.5 miles of perennial stream, owing to the prohibitive cost of cruising short reaches. This restriction removed 36 miles of stream from the sampling pool. Another 10 miles of stream in the Idaho land base were excluded due to logistical constraints. This left 259 sections of land encompassing 405 miles (80%) of perennial streams in Tier I basins available to sample. Sixty-four of these 259 sections were chosen using systematic random sampling.

1.2 Sampling Intensity

Within each chosen section, ten sample sites were located along perennial streams at seven chain (462 ft) intervals. At each site, two fixed-area circular plots were established. One plot was located 26.5 from the stream bank and the other at 79 ft. These distances were chosen to coincide with Montana's 50 ft stream buffers, and an approximate site potential tree height of 100 ft for sites east of the Cascade Crest (Arno et al. 1985, Pfister et al. 1977). Both plots were placed on the same line running perpendicular to the stream channel. Using this sampling intensity, 1,280 plots were measured, each representing 8.1 acres of land in riparian areas along perennial streams within Tier I basins.

1.3 Cruise Methods

One of four plot sizes (1/20, 1/40, 1/70, and 1/100 ac) was chosen by the cruiser to measure each plot. Plot size was judged to be adequate when at least seven trees of the main stand fell within the

31 Watersheds that contain streams known to be important for bull trout spawning and juvenile rearing

plot boundaries. This minimum tree count was not required if the largest (1/20 ac) plot size was measured. On each plot all live trees taller than 4.5 feet (i.e., diameter breast height; dbh) were tallied by species and 1-inch dbh class. At least one tree on every plot was measured for height, and all others were estimated to the nearest ten feet. Dead trees larger than 4.5-inch dbh were also tallied within each plot.

1.4 Cruise Summary

Cruise data were first converted to per acre values for common reference. Each cruised plot was then assigned a forest type based upon its measured attributes. The number of perennial stream miles within a given forest type was then computed using simple proportional allocation (Thompson 1992).

1.5 Summary of Riparian Stand Characteristics

The sampled riparian stands were classified into nine separate groups based on tree size (quadratic mean diameter at 4.5 ft, QMD) and density (trees per acre, TPA). Table C-1 identifies the values of QMD & TPA for each riparian forest stand type, and Table C-2 provides labels for each type. In addition to the forest types listed in the tables there is a non-stocked class representing plots without trees.

Table C-1. Characteristics of dominant riparian stand types on Plum Creek lands in the interior Columbia River Basin.

	Small (3) 0"<QMD≤6"	Medium (9) 6"<QMD ≤12"	Large (15) 12"<QMD≤18"	Huge (20) QMD >18"
Low Density (L)	= 300 TPA	=200 TPA	=150 TPA	=150 TPA
High Density (H)	300 >TPA = 700	200>TPA= 500	150>TPA= 300	150>TPA= 300
Thick Density (T)	TPA > 700	TPA > 500	TPA > 300	TPA > 300

Table C-2. Labels for dominant riparian stand types on Plum Creek lands in the interior Columbia River Basin.

	Small (3)	Medium (9)	Large (15)	Huge (20)
Low Density (L)	L3	L9	L15	L20
High Density (H)	H3	H9	H15	H20
Thick Density (T)	T3	T9	T15	T20

Each of the nine dominant stand types[32] exhibits distinctive "average" characteristics described in terms of tree density, size, and species. Each has different growth trajectories, and therefore different harvest or enhancement opportunities. Detailed descriptions of the characteristics and abundance of each dominant stand type are provided below along with visual representations created by the Stand Visualization System (Magaughey *in prep*).

1.6 Forest stand visualization system software

The Stand Visualization System (SVS) was developed by the USDA-Forest Service (Magaughey *in prep,* Magaughey 1997). Inventory data input to the Forest Vegetation Simulator (FVS) can be used to accurately portray outcomes of a various structural trajectories (Stage 1973, Wykoff et al. 1983). SVS accepts data from a tree list, such as output from the FVS, and assigns a spatial arrangement to the trees. The representation of the trees requires at a minimum species, DBH, height, and LCR. Spatial arrangement can be controlled by designating random or systematic distribution of tree location.

32 The "Huge" category in Tables C-1 and C-2 was not included in the analysis, because the total combined area for these stands represented less than 2% of the streamside miles in the project area.

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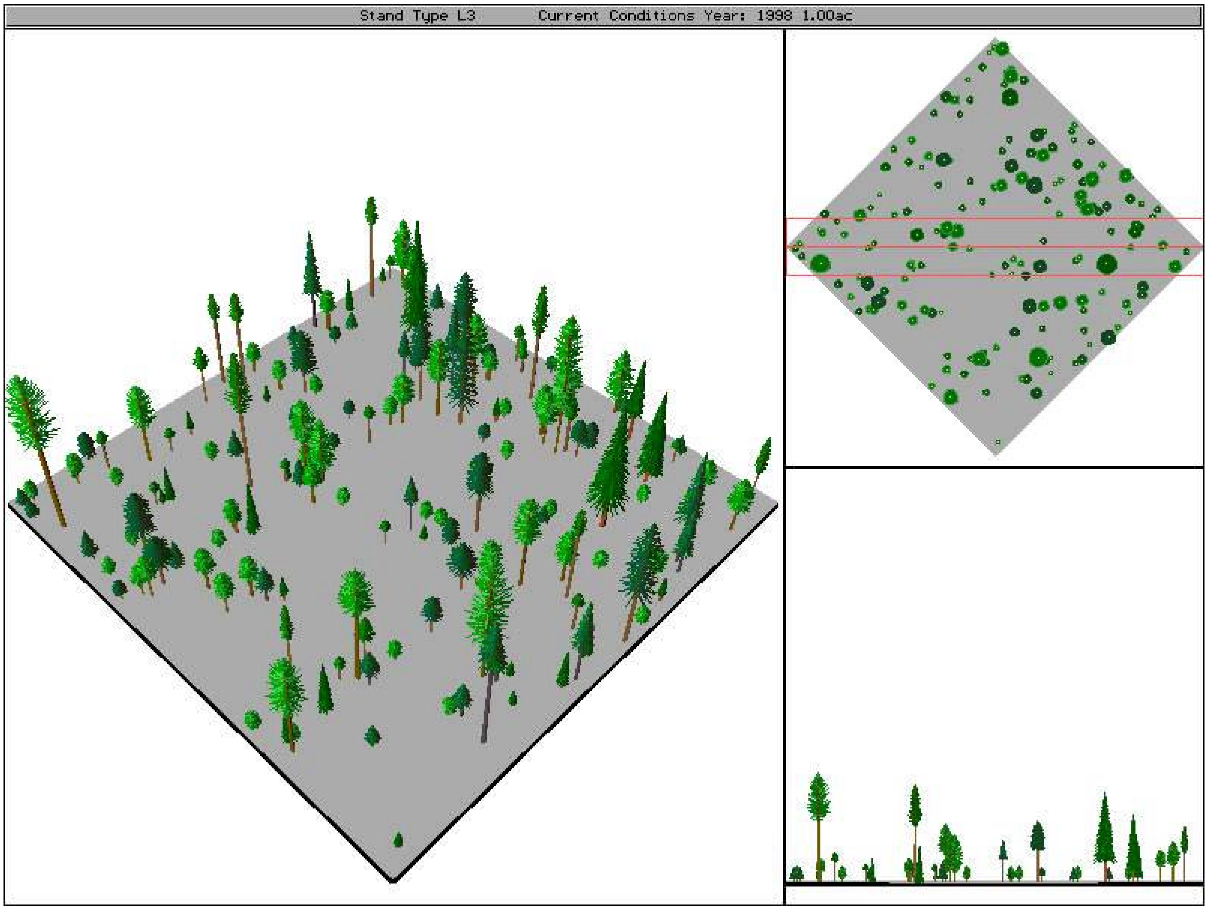


Riparian Stand Type L3

- **Summary Description**
  Low density stands of medium-sized trees.

- **Tree Density**
  Less than 300 trees per acre.

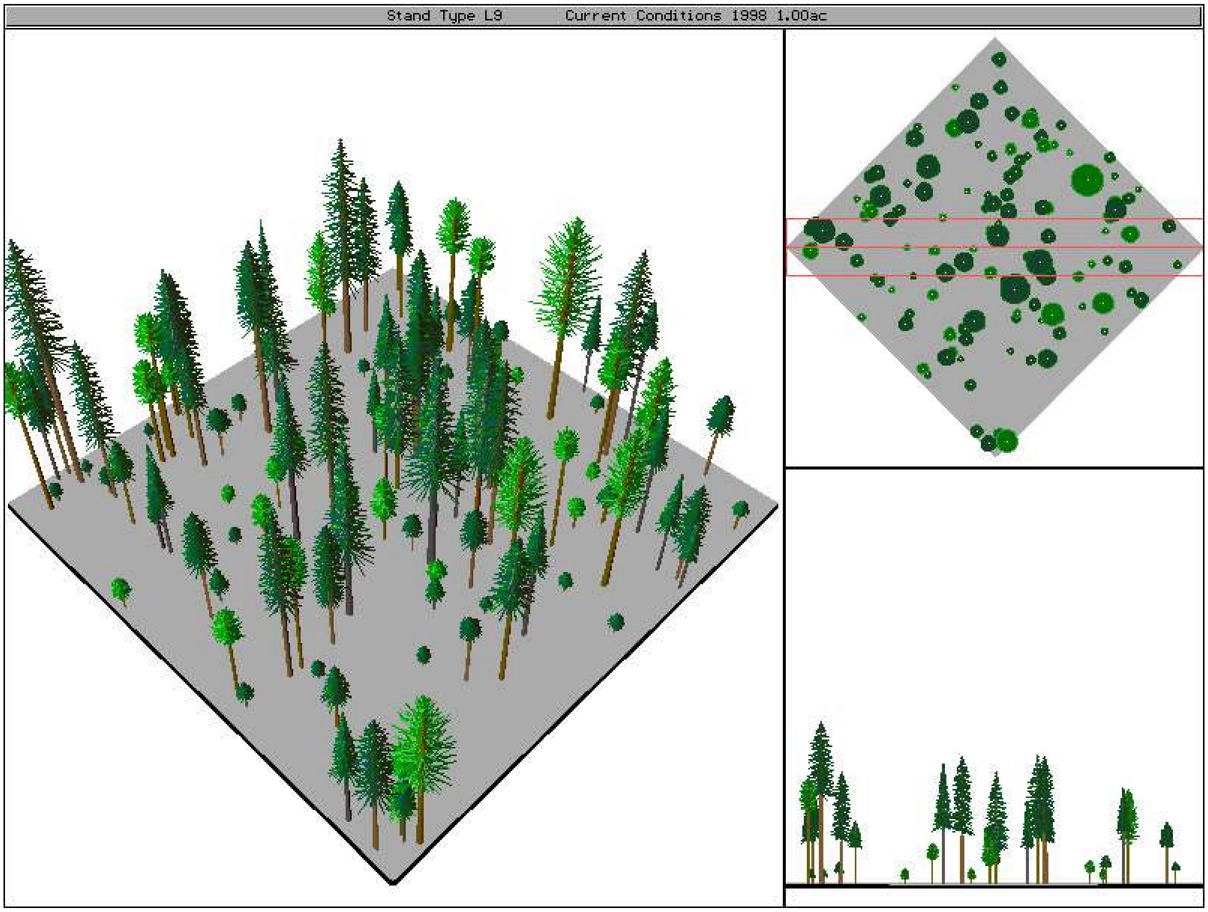
- **Tree Sizes**
  Mean Diameter between 0” and 6”

- **Tree Species Mix**
  Generally a mixture of Douglas Fir, Alpine Fir, Lodgepole Pine, and Englemann Spruce in Montana. In Washington, Silver Fir often replaces the Englemann Spruce component. Species mixtures can run from nearly pure species to very heterogeneous. Associated minor species include Western Larch, Grand Fir, Ponderosa Pine, Mountain Hemlock, White Pine, and Cottonwood.

- **Percent of Total Riparian Forest in the Project Area**
  13.4% ± 2.6%

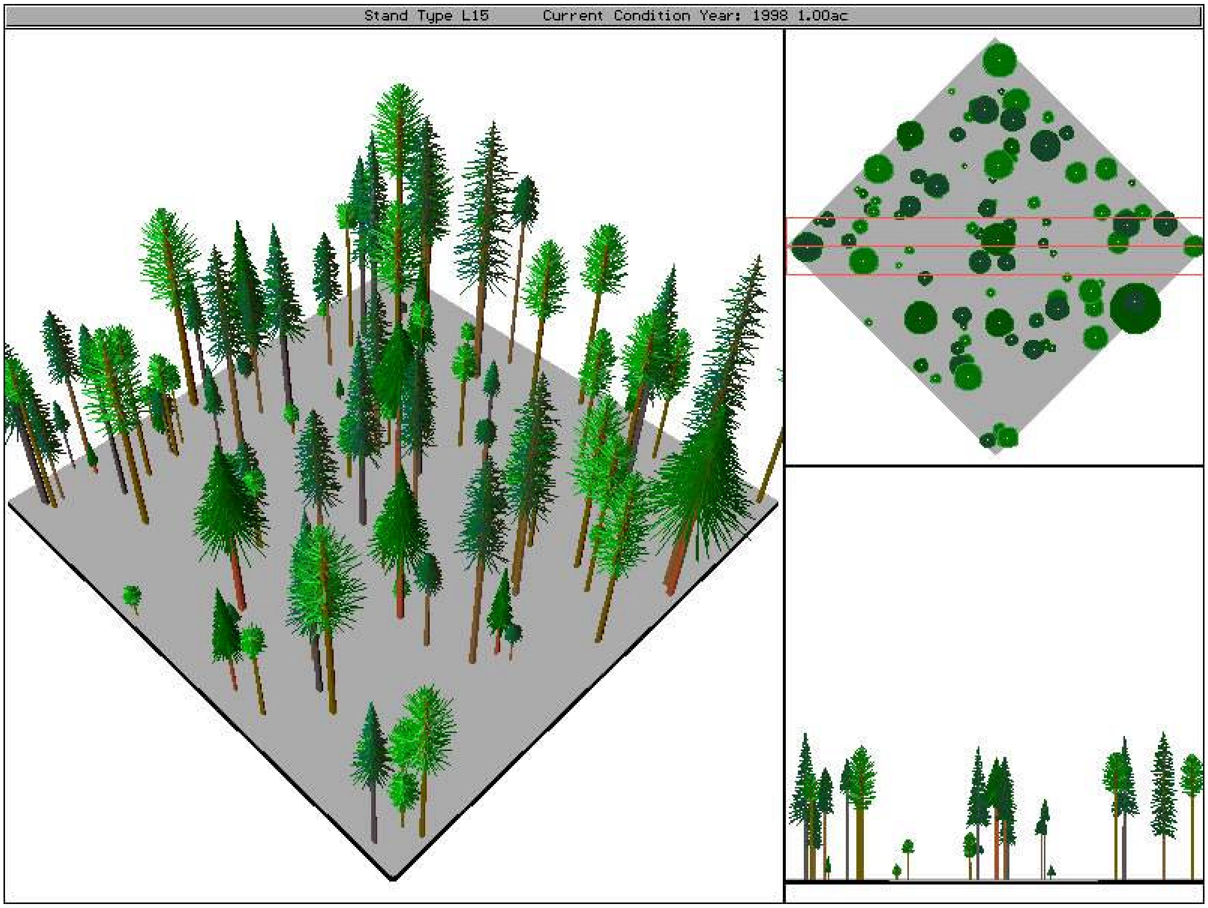
- **Harvest Opportunity**
  Long-term harvest opportunities with possible limited near-term opportunities depending upon stand structure.

- **Silvicultural Opportunities**
  Limited single tree selection in all-aged/multi-storied stands.



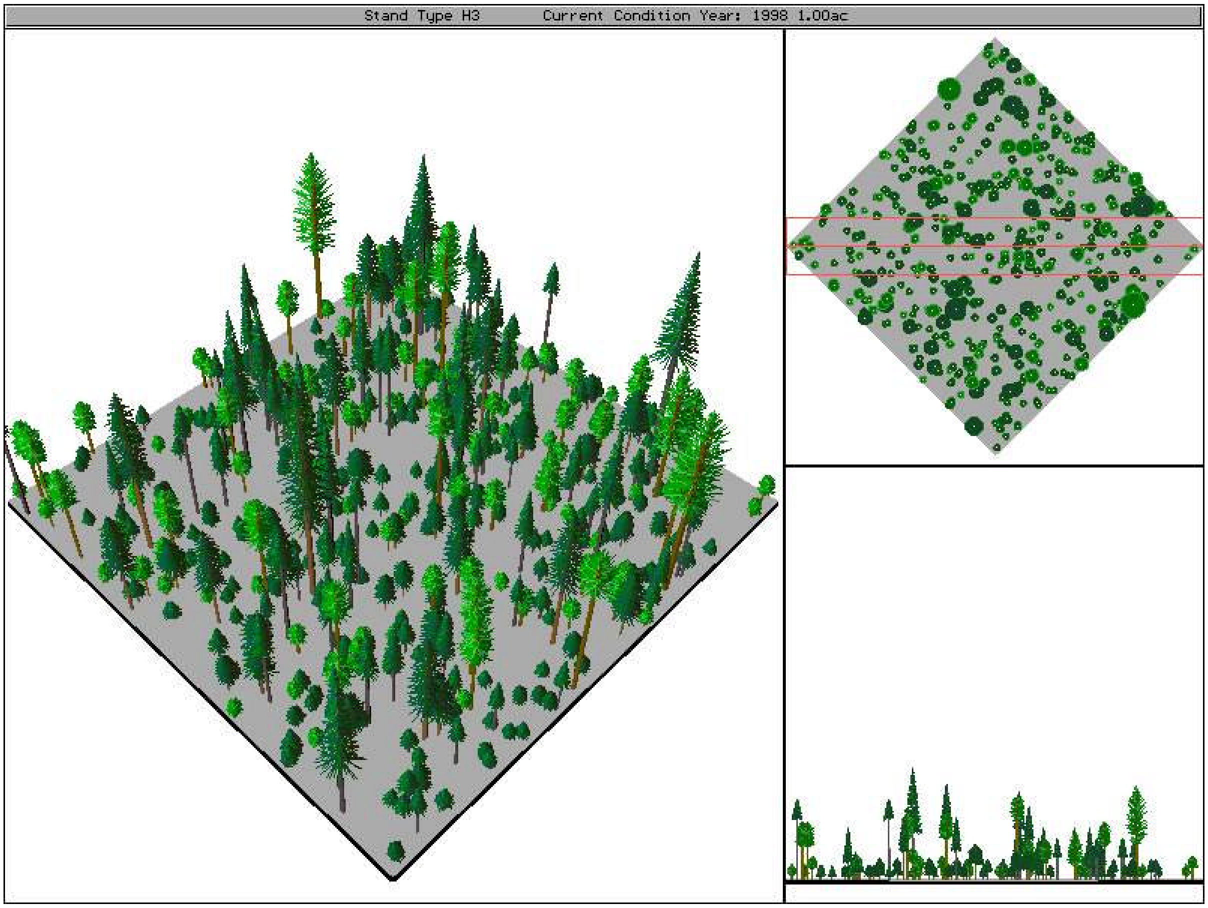
Riparian Stand Type L9

- **Summary Description**
  Low density stands of medium-sized trees.
- **Tree Density**
  Less than 200 trees per acre.
- **Tree Sizes**
  Mean Diameter between 6" and 12"
- **Tree Species Mix**
  Generally a mixture of Douglas Fir, Alpine Fir, Grand Fir, and Englemann Spruce in Montana. In Washington, Silver Fir often replaces the Englemann Spruce component. Species mixtures can run from nearly pure species to very heterogeneous. Associated minor species include Western Larch, Lodgepole Pine, Ponderosa Pine, Mountain Hemlock, White Pine, and Cottonwood.
- **Percent of Total Riparian Forest in the Project Area**
  9.8% ± 2.2%
- **Harvest Opportunity**
  Current and near-term harvest opportunities are very limited due to existing SMZ laws.
- **Silvicultural Opportunities**
  Single tree selection in all-aged/multi-storied stands. Commercial thinning opportunities in even-aged/single storied stands.



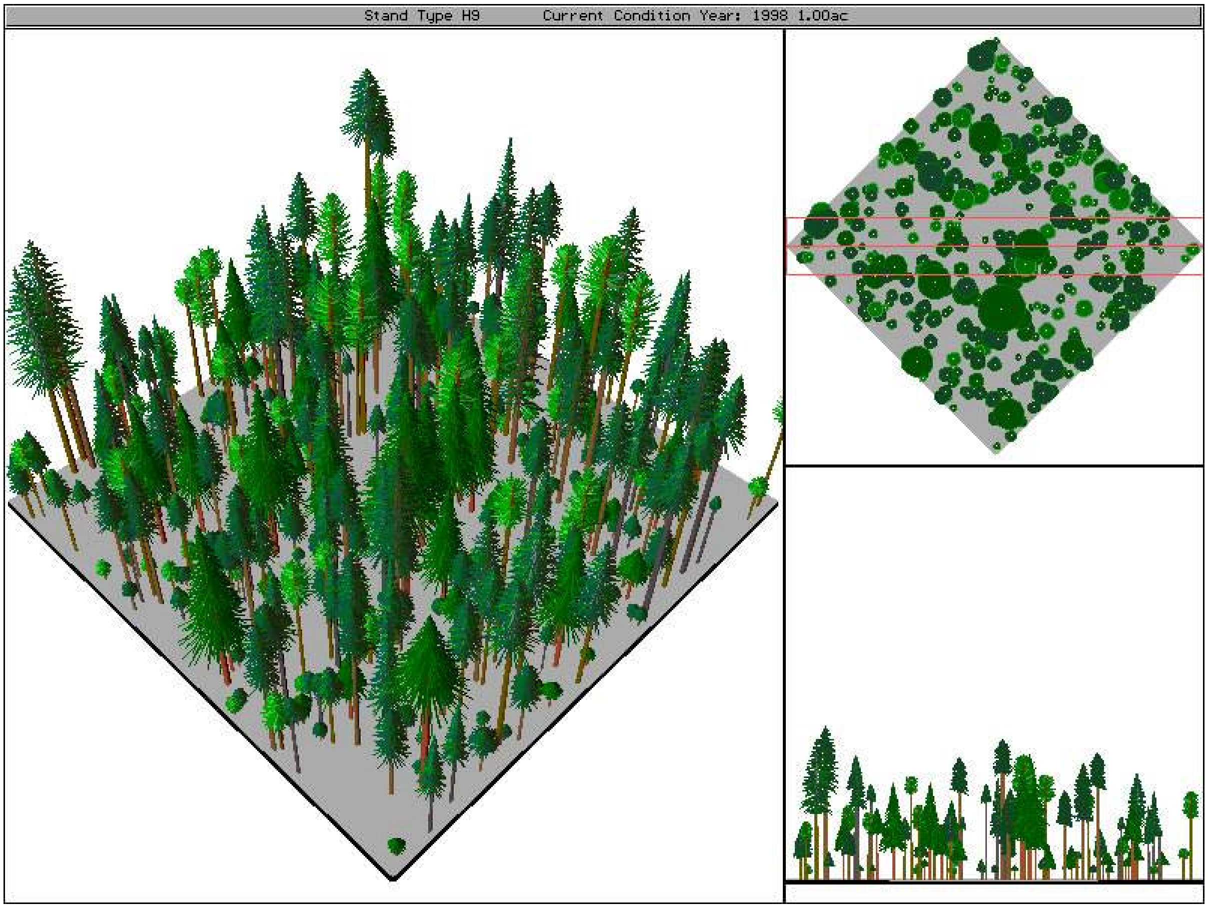
Riparian Stand Type L15

- **Summary Description**
  Low density stands of large-sized trees.
- **Tree Density**
  Less than 150 trees per acre.
- **Tree Sizes**
  Mean Diameter between 12" and 18"
- **Tree Species Mix**
  Generally a mixture of Douglas Fir, Alpine Fir, Grand Fir, and Englemann Spruce in Montana. In Washington, Silver Fir often replaces the Englemann Spruce component. Species mixtures can run from nearly pure species to very heterogeneous. Associated minor species include Western Larch, Lodgepole Pine, Ponderosa Pine, Mountain Hemlock, White Pine, and Cottonwood.
- **Percent of Total Riparian Forest in the Project Area**
  4.5% ± 1.6%
- **Harvest Opportunity**
  Current harvest opportunities are very limited due to existing SMZ laws.
- **Silvicultural Opportunities**
  Single tree selection in all-aged/multi-storied stands and shelterwood cutting to promote stand regeneration.



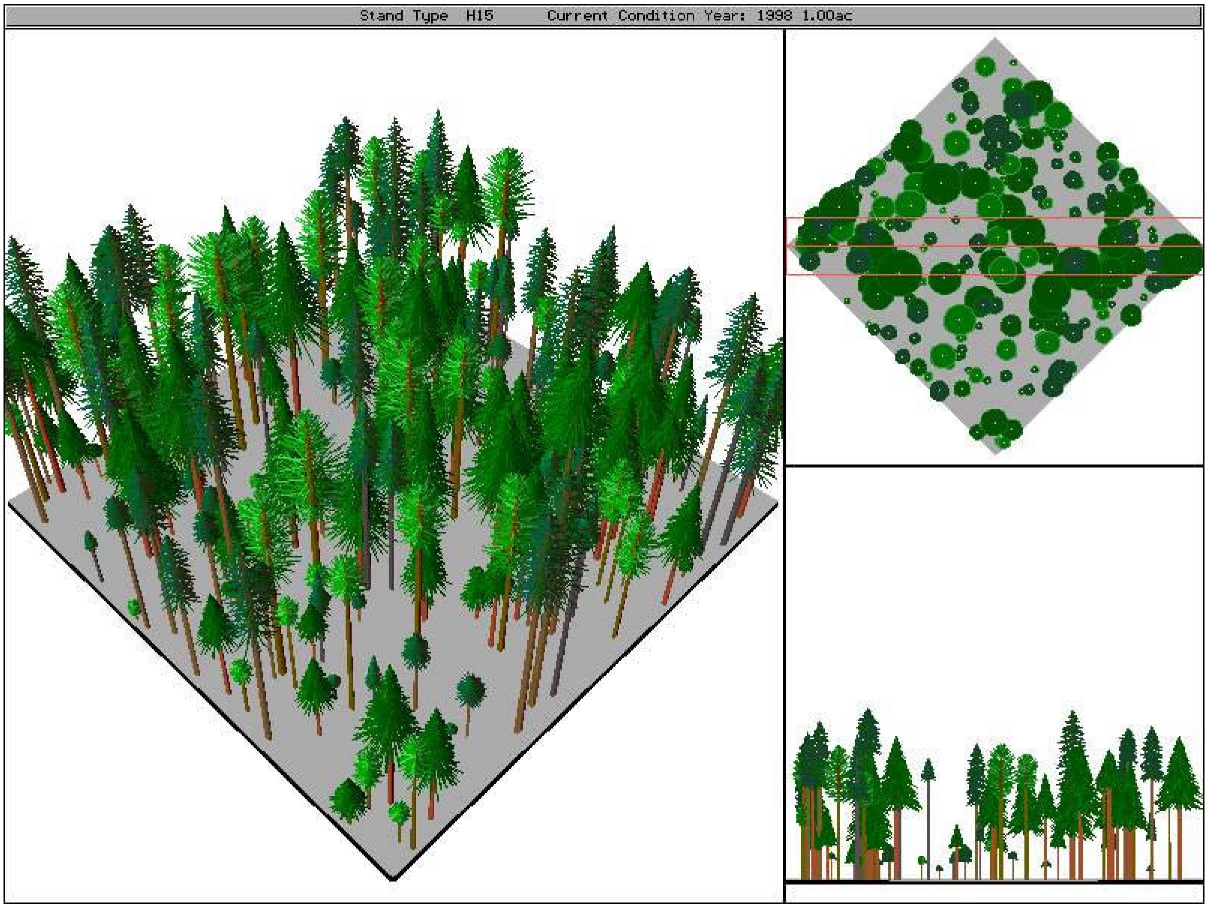
Riparian Stand Type H3

- **Summary Description**
  Dense stands of small-sized trees.
- **Tree Density**
  300 – 700 trees per acre.
- **Tree Sizes**
  Mean Diameter between 0” and 6”
- **Tree Species Mix**
  Generally a mixture of Douglas Fir, Alpine Fir, Western Red Cedar, Grand Fir, and Englemann Spruce in Montana. In Washington, Silver Fir often replaces the Englemann Spruce component. Species mixtures can run from nearly pure species to very heterogeneous. Associated minor species include Western Larch, Lodgepole Pine, Mountain Hemlock, White Pine, and Cottonwood.
- **Percent of Total Riparian Forest in the Project Area**
  13.3% ± 2.5%
- **Harvest Opportunity**
  Mostly long-term opportunities with near-term opportunities possible in some stands depending upon stand structure.
- **Silvicultural Opportunities**
  Pre-commercial thinning opportunities in even-aged/single-storied stands. Single tree selection in all-aged/multi-storied stands.



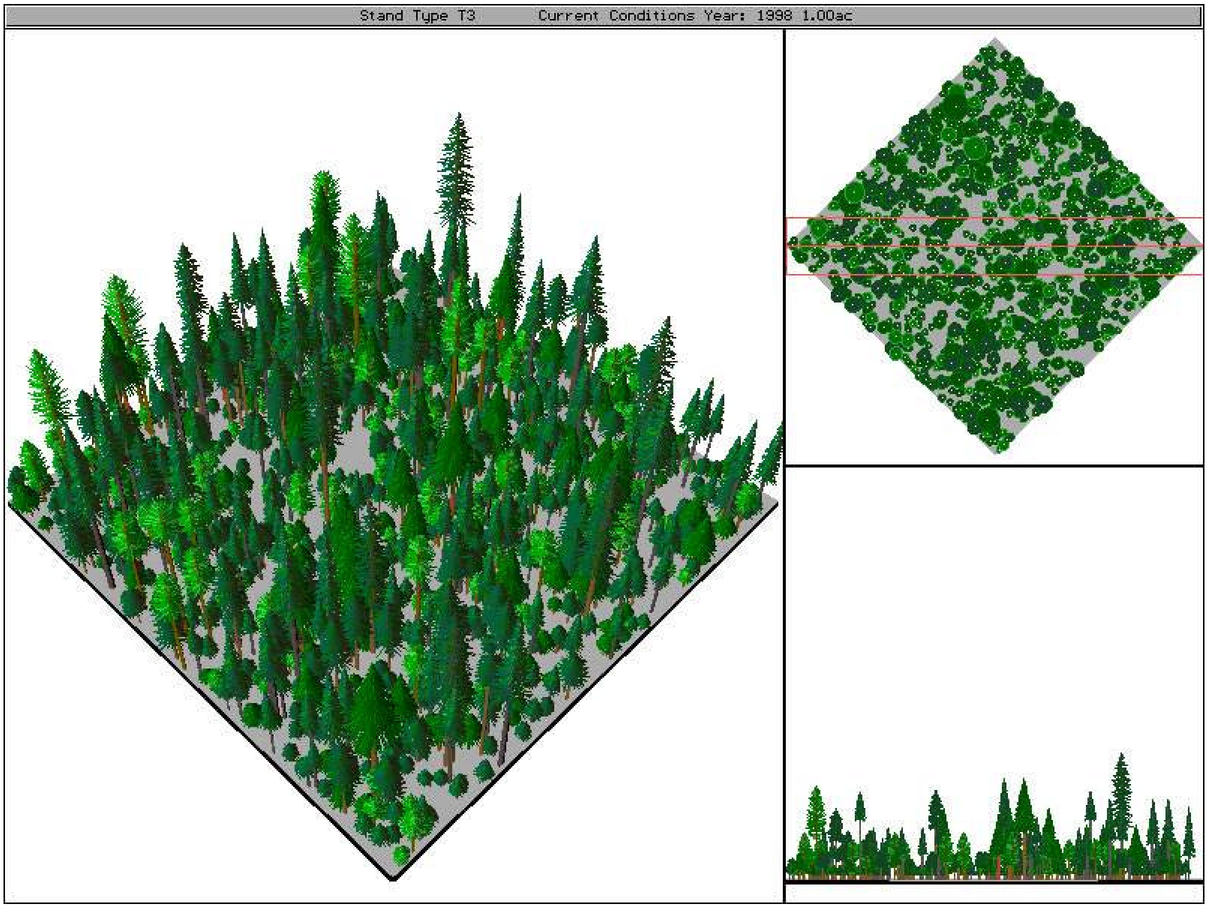
Riparian Stand Type H9

- **Summary Description**
  Dense stands of medium-sized trees.
- **Tree Density**
  200 – 500 trees per acre.
- **Tree Sizes**
  Mean Diameter between 6” and 12”
- **Tree Species Mix**
  Generally a mixture of Douglas Fir, Alpine Fir, Western Red Cedar, Grand Fir, and Englemann Spruce in Montana. In Washington, Silver Fir often replaces the Englemann Spruce component. Species mixtures can run from nearly pure species to very heterogeneous. Associated minor species include Western Larch, Lodgepole Pine, Mountain Hemlock, White Pine, and Cottonwood.
- **Percent of Total Riparian Forest in the Project Area**
  23.4% ± 3.2%
- **Harvest Opportunity**
  Current to near-term opportunities may exist depending upon stand structure.
- **Silvicultural Opportunities**
  Single tree selection in all-aged/multi-storied stands. Commercial thinning opportunities in even-aged/single-storied stands.



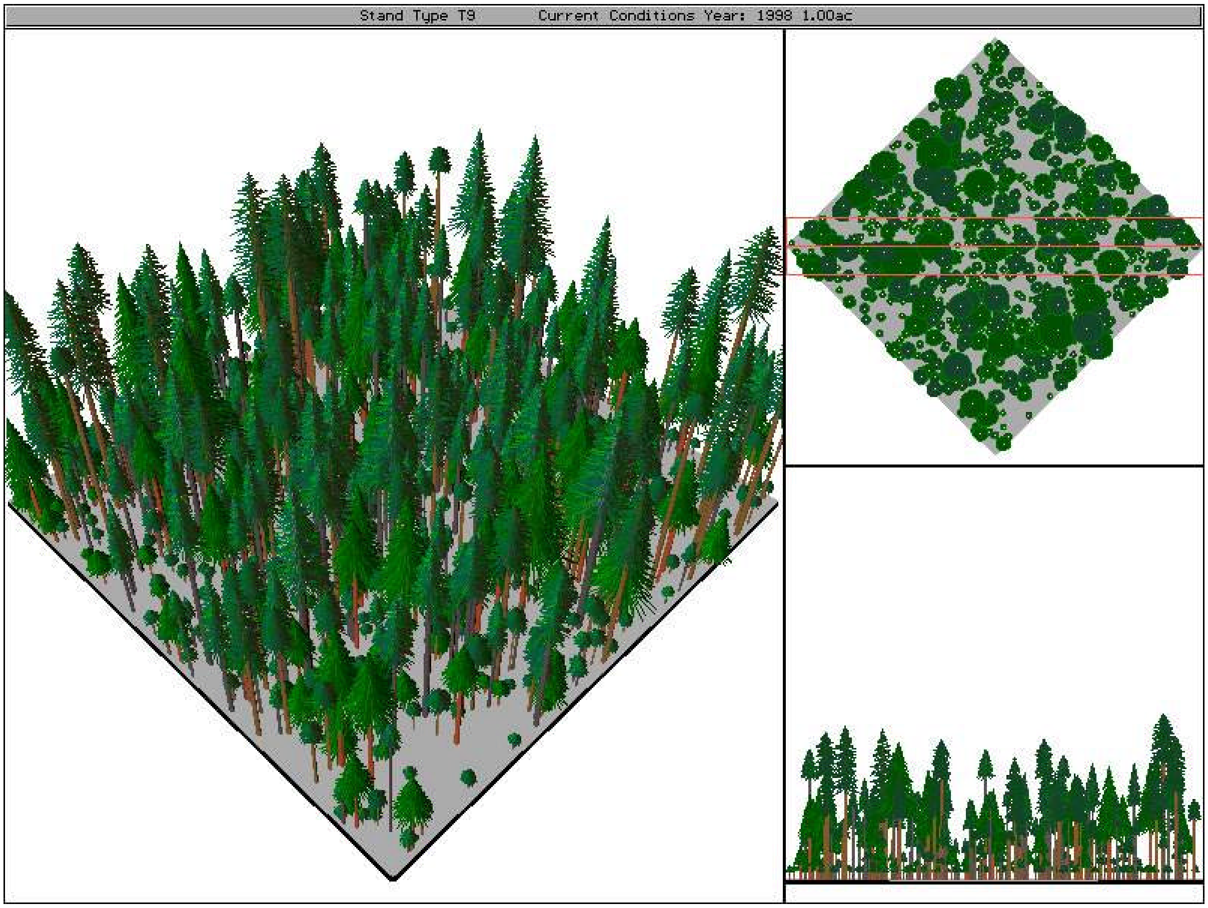
Riparian Stand Type H15

- **Summary Description**
  Dense stands of large-sized trees.
- **Tree Density**
  150 – 300 trees per acre.
- **Tree Sizes**
  Mean Diameter between 12" and 18"
- **Tree Species Mix**
  Generally a mixture of Douglas Fir, Alpine Fir, Western Red Cedar, Grand Fir, and Englemann Spruce in Montana. In Washington, Silver Fir often replaces the Englemann Spruce component. Species mixtures can run from nearly pure species to very heterogeneous. Associated minor species include Western Larch, Lodgepole Pine, Mountain Hemlock, White Pine, and Cottonwood.
- **Percent of Total Riparian Forest in the Project Area**
  3.9% ± 1.4%
- **Harvest Opportunity**
  Current harvest opportunities.
- **Silvicultural Opportunities**
  Single tree selection in all-aged/multi-storied stands.



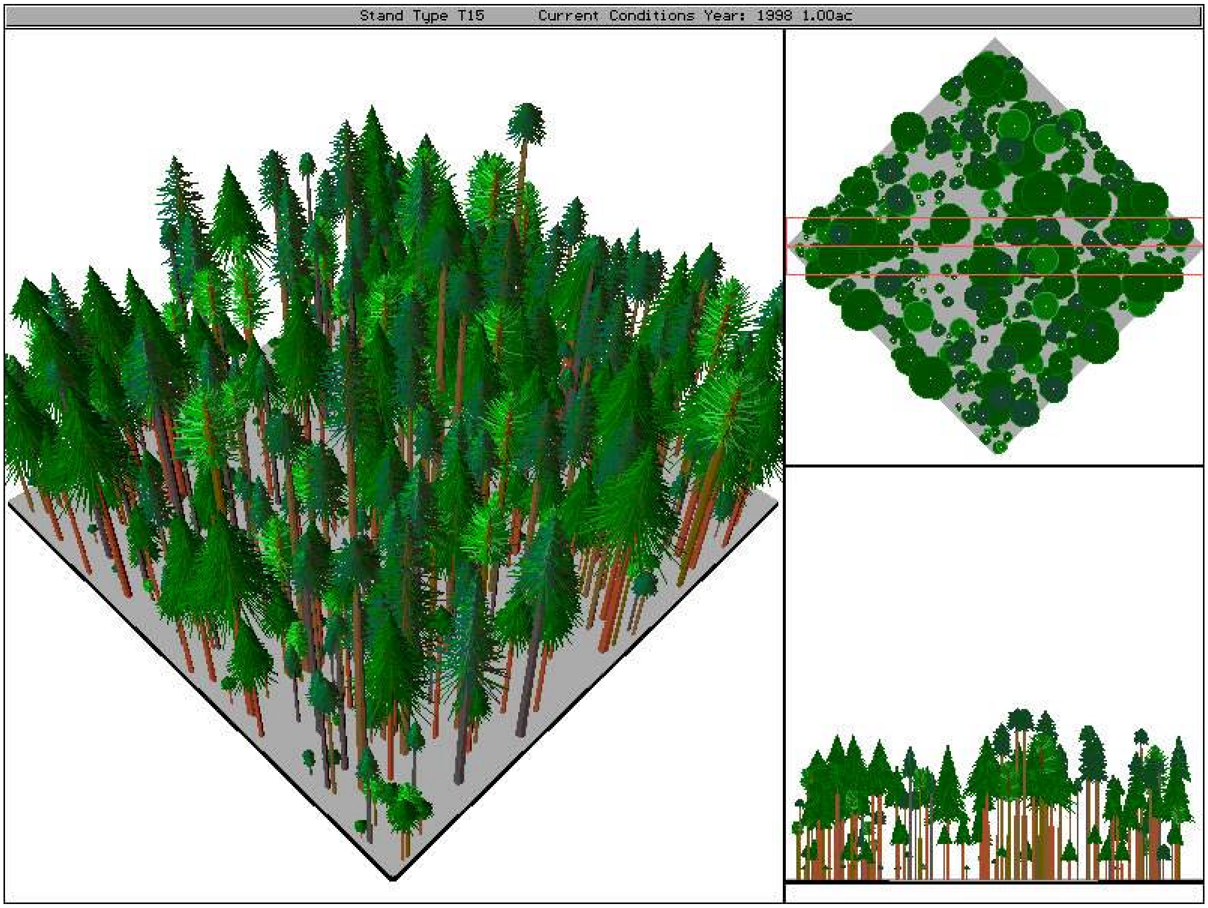
Riparian Stand Type T3

- **Summary Description**
  Very thick stands of small-sized trees.
- **Tree Density**
  More than 700 trees per acre.
- **Tree Sizes**
  Mean Diameter between 0" and 6"
- **Tree Species Mix**
  Generally a mixture of Douglas Fir, Alpine Fir, Western Red Cedar, Grand Fir, and Englemann Spruce in Montana. In Washington, Silver Fir often replaces the Englemann Spruce component. Species mixtures can run from nearly pure species to very heterogeneous. Associated minor species include Western Larch, Lodgepole Pine, Mountain Hemlock, White Pine, and Cottonwood.
- **Percent of Total Riparian Forest in the Project Area**
  12.8% ± 2.5%
- **Harvest Opportunity**
  Limited near-term harvest opportunities may exist depending upon stand structure.
- **Silvicultural Opportunities**
  Pre-commercial and some commercial thinning for pulpwood. Single tree selection in all-aged/multi-storied stands with near-term harvest opportunities.



Riparian Stand Type T9

- **Summary Description**
  Very thick stands of medium-sized trees.
- **Tree Density**
  More than 500 trees per acre.
- **Tree Sizes**
  Mean Diameter between 6" and 12"
- **Tree Species Mix**
  Generally a mixture of Douglas Fir, Alpine Fir, Western Red Cedar, Grand Fir, and Englemann Spruce in Montana. In Washington, Silver Fir often replaces the Englemann Spruce component. Species mixtures can run from nearly pure species to very heterogeneous. Associated minor species include Douglas Fir, Grand Fir, Western Larch, Lodgepole Pine, Mountain Hemlock, White Pine, and Cottonwood.
- **Percent of Total Riparian Forest in the Project Area**
  8.4% ± 2.1%
- **Harvest Opportunity**
  Current and near-term harvest opportunities exist depending upon stand structure.
- **Silvicultural Opportunities**
  Commercial thinning in even-aged stand. Single tree selection in all-aged/multi-storied stands.



Riparian Stand Type T15

- **Summary Description**
  Very thick stands of large-sized trees.
- **Tree Density**
  More than 500 trees per acre.
- **Tree Sizes**
  Mean Diameter between 12" and 18"
- **Tree Species Mix**
  Generally a mixture of Douglas Fir, Alpine Fir, Western Red Cedar, Grand Fir, and Englemann Spruce in Montana. In Washington, Silver Fir often replaces the Englemann Spruce component. Species mixtures can run from nearly pure species to very heterogeneous. Associated minor species include Grand Fir, Western Larch, Lodgepole Pine, Mountain Hemlock, White Pine, and Cottonwood.
- **Percent of Total Riparian Forest in the Project Area**
  4.4% ± 1.5%
- **Harvest Opportunity**
  Current and near-term harvest opportunities exist depending upon stand structure.
- **Silvicultural Opportunities**
  Commercial thinning in even-aged stand. Single tree selection in all-aged/multi-storied stands.